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Technical and economic study on mercury emission control technologies for combustion power plants

Thesis submitted in partial fulfilment of the requirements for the degree of Master of Science in Technology.

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Diplomityön tiivistelmä

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Tiivistelmä

Työssä tutkittiin erilaisten savukaasunpuhdistustekniikoiden soveltuvuutta ja kustannustehokkuutta polttovoimalaitosten elohopeanpoistossa. Työ tehtiin teollisuuspäästödirektiiviin (Industrial Emissions Directive, IED) liittyvän BREF-asiakirjan asettamien elohopeapäästöraja-arvojen puitteissa. Päätavoitteena oli selvittää näiden raja-arvojen saavuttamisesta koituvat kustannukset.

Työn kirjallisuusosuudessa luotiin katsaus elohopean aiheuttamaan saastumiseen, polttoaineiden elohopeapitoisuuteen, maapallon elohopeapäästöihin ja elohopeapäästöjä koskevaan sääntelyyn. Tutkimusosassa tutkittiin erilaisten savukaasunpuhdistustekniikoiden elohopeanpoistokyky ja kustannusanalyysin perustana olevat päästöt. Tämän jälkeen selvitettiin tarve elohopean lisäpoistolle ja tästä aiheutuvat kustannukset. Kustannukset laskettiin olemassa oleville hiilivoimalaitoksille erilaisilla savukaasunpuhdistuslaitteistoilla. Tutkitut voimalaitoskoot olivat sähköteholtaan 100, 250 ja 500 MW_e. Laskelmat tehtiin päästörajoille 9, 4 and 1 $\mu g/Nm^3$. Laskelmissa käytettiin puolalaisen bitumisen kivihiilen ominaisuuksia. Analyysi tehtiin kolmelle elohopeanpoistotekniikalle, jotka olivat tavanomaisen aktiivihiilen injektointi, brominoidun aktiivihiilen injektointi ja bromin lisäys. Kustannusarvot perustuvat kirjallisuudesta ja teknologiatoimittajilta saatuihin tietoihin.

Työn aikana selvisi, että huomattavia määriä elohopeaa on mahdollista poistaa jo olemassa olevilla savukaasunpuhdistuslaitteilla, kun käytetään bitumista kivihiiltä. Joissain tapauksissa alkuperäinen elohopeanpoistokyky voi olla riittävä saavuttamaan tiukimmatkin raja-arvot. Lisäpoistoa vaativissa tapauksissa kustannusten laskettiin olevan 0,034 – 0,697 €/MWh_{th} tai 18 000 – 374 400 €/kg, riippuen laitoksen koosta, olemassa olevasta laitteistosta, raja-arvosta ja käytetystä lisäpoistotekniikasta. Bromin lisäys havaittiin olevan edullisin tekniikka, jota seurasi brominoidun aktiivihiilen ja tavanomaisen aktiivihiilen injektointi. Elohopeanpoiston kustannukset ovat kuitenkin hyvin voimalaitoskohtaisia. Vahvasti kustannuksiin vaikuttavia laitoskohtaisia tekijöitä ovat muun muassa polttoaineen ja polttokattilan ominaisuudet sekä olemassa olevan puhdistuslaitteiston suorituskyky.

Avainsanat elohopea, elohopean poisto, saasteenesto, elohopeanpoistotekniikat, elohopeapäästöt, polttovoimalaitos, taloudellinen tarkastelu, teollisuuspäästödirektiivi, hiili, kustannusanalyysi, kustannukset

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Abstract

This study explored the applicability and cost-efficiency of current flue gas cleaning technologies for mercury emission control in combustion power plants. The study was made in the context of mercury emission limits set by the BREF-document which is related to the Industrial Emissions Directive (IED). The main goal was to figure out the costs that are occurred when these limits are achieved.

The literature part of the study gives a review on mercury pollution, mercury content in fuels, worldwide mercury emissions and mercury emission regulation. The research part involves definition of baseline mercury removal efficiency of different flue gas cleaning devices and baseline mercury emissions across the studied configurations. From that basis, the need for additional mercury removal was determined and the costs of the removal were estimated. The cost analysis was made for existing coal-fired power plants that had different flue gas cleaning configurations. The studied plant capacities were 100, 250 and 500 MW_e, and the applied emission limits were 9, 4 and 1 $\mu g/Nm^3$. Properties of Polish bituminous hard coal were used. The analysis was limited to three most promising mercury-specific removal technologies: untreated activated carbon injection, brominated activated carbon injection and bromine injection. The cost estimation is based on values found in literature and data received from technology suppliers.

The study discovered that significant amounts of mercury can be removed by the existing flue gas cleaning devices alone, when bituminous coal is used. In some cases, the existing equipment may be sufficient to meet even the strictest emission levels. For cases that required additional removal, the annual costs were determined to be in the range of 0,034 – 0,697 €/MWh_{th} or 18 000 – 374 400 €/kg Hg, depending on the plant size, emission limit, existing equipment and applied mercury removal technology. Bromine injection was discovered to be the cheapest technology, followed by brominated activated carbon injection and untreated activated carbon injection. However, it became also clear that the costs of mercury removal are very plant-specific, depending on factors like fuel properties, boiler conditions and the performance of existing flue gas cleaning equipment.

Keywords mercury, mercury removal, emission control, removal technologies, mercury emissions, combustion power plant, Industrial Emissions Directive, coal, cost analysis, costs

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Ville Laurén

Contents

List of abbreviations

ACI	activated carbon injection
APCD	air pollution control device
BAT	best available technique
BAT-AEL	emission levels associated with the best available techniques
BREF	BAT reference document
CFB	circulating fluidized bed
DOE	Department of Energy (U.S.)
EPA	Environmental Protection Agency (U.S.)
EPRI	Electric Power Research Institute (U.S.)
E-PRTR	European Pollutant Release and Transfer Register
ESP	electrostatic precipitator
FF	fabric filter
FGD	flue gas desulphurization
IED	Industrial Emission Directive
LCP	Large Combustion Plant (Directive)
LOI	loss on ignition
NETL	National Energy Technology Laboratory (U.S)
NID	Novel Integrated Desulphurization
PAC	powdered activated carbon
PJFF	pulse jet fabric filter
PRB	Powder River Basin (Coal type)
PM	particulate matter
SCR	selective catalytic reduction
SDA	semi-dry absorber
TOXECON	toxic emission control process
UBC	unburned carbon
UNECE	United Nations Economic Commission for Europe
UNEP	United Nations Environment Programme
WFGD	wet flue gas desulphurization

List of chemical symbols

Br_2	bromine (molecule)
Cl_2	chlorine (molecule)
$CaBr_2$	calcium bromide
$CaCl_2$	calcium chloride
$CaCO_3$	calcium carbonate (limestone)
CaO	calcium oxide (lime)
$Ca(OH)_2$	calcium hydroxide (slaked lime)
$CaSO_3$	calcium sulphite
$CaSO_4$	calcium sulphate
CO	carbon monoxide
CO_2	carbon dioxide
HBr	hydrogen bromide
HCl	hydrogen chloride
Hg^{2+}	oxidized form of mercury
Hg^0	elemental form of mercury
$Hg(p)$	particulate-bound form of mercury
HgS	mercury sulphide (cinnabar)
HNO_3	nitric acid
H_2SO_4	sulphuric acid
$NaBr$	sodium bromide
$NaHS$	sodium hydrosulphide
NH_3	ammonia
$(NH_4)_2SO_4$	ammonium sulphate
NO	nitrogen oxide
NO_2	nitrogen dioxide
O_2	oxygen (molecule)
O_3	ozone
SO_2	sulphur dioxide
SO_3	sodium trioxide

1 Introduction

Mercury is a highly toxic pollutant that has adverse effects on humans and wildlife. Mercury is especially problematic because of its persistent nature and ability to long-range transport, which enable it to cycle globally through the atmosphere. As a basic chemical element, mercury cannot be destroyed nor degraded into something else. Instead, once released into the environment, it slowly accumulates in living organisms. Anthropogenic actions increase the amount of mercury in the global cycle, and since the Industrial Revolution, the mercury content in the environment has been clearly increasing. This has raised concern over the harmful effect of mercury.

A significant amount of the anthropogenic mercury emissions are originating from combustion power plants, mostly coal-fired plants. Although mercury air emissions are decreasing in North America and Europe due to stringent emission regulation and wide-spread use of flue gas cleaning devices, emissions keep increasing especially in South-East Asia. The reason for this is the rapid industrialization and increase in energy demand, which is satisfied mostly by coal-fired power plants.

In order to address the mercury problem, several international conventions and national legislation measures have been implemented. In the European Union, the main emission control instrument for large power plants is Industrial Emission Directive (IED). An essential part of the IED is a legally binding Best Available Techniques Reference document (BREF), which sets emission limits for different pollutants. Currently, a new revision of the BREF document is under preparation and is likely to be published during the year 2016. The revised BREF will tighten the emission limits for most common pollutants, such as nitrogen oxide, sulphur dioxide and dust particles. Moreover, emission limits are set for new pollutant components, also for mercury. The new emission limits will probably come into force in the beginning of 2018, after which there is 4 years transition period for power plant operators and other parties to adopt them.

Because emission limits will be also set for mercury, power plant operators have to pay more attention to mercury control. Operators have to follow their mercury emissions and take actions to reduce the emissions, if needed. During the upcoming years, there will likely be a growing interest towards mercury removal technologies and the costs of mercury removal.

Currently, varying amounts of mercury are removed in power plants indirectly by conventional emission control technologies that are mainly used for other pollutants, such as sulphur dioxide, nitrogen oxides and particulate matter. These technologies include flue gas desulphurization, selective catalytic reduction, electrostatic precipitators and fabric filters. For cases where mercury removal is not sufficient with the conventional technologies, advanced mercury-specific technologies have been developed, including injection of different sorbents, such as activated carbon, and addition of halogen-based chemicals. These technologies are meant to be used in conjunction with conventional flue gas cleaning technologies in order to increase the mercury removal efficiency.

When a power plant operator considers implementation of mercury-specific removal technology, there are several factors to be considered. These are mercury content and chemical composition of the fuel, boiler type, combustion conditions and existing air pollution control devices and their performance. Because of the uncertainties, it is not an easy task for a power plant operator to choose the right mercury removal technology. Consequently, the costs of the new technology are difficult to assess.

Although some cost-assessments have been made for mercury removal, they have mostly taken place in the USA. There are not many cost assessments that have been made for European power plants within the context of the emission limits of the BREF document. Thus, it is unclear whether the co-benefit removal efficiency of the existing flue gas cleaning devices is sufficient to achieve the required emission limits and if not, how much additional mercury removal is needed. Furthermore, the costs of additional removal remain unknown.

1.1 Objectives of the work

The main objective of this thesis is to figure out applicability and costs of the most common mercury emission control technologies for combustion power plants in the context of the BREF document. The purpose is to determine the costs of additional removal that is needed to achieve the new emission limits. The scope is combustion power plants within the European Union, where the BREF document is applied. The focus will be in coal-fired power plants. Cost analysis will be carried out for existing power plants that already have some kind of flue gas cleaning devices.

The study is divided in two main parts, literature part and research part. The main purpose of the literature part is to introduce technologies used for mercury removal and determine their current status in combustion power plants. Another important objective is to clarify the legislation around mercury control and explain the mercury emission levels that are applied in this study. The literature part gives also an introduction on global mercury pollution and behaviour of mercury in combustion power plants. Furthermore, mercury content of coal, peat and oil are examined.

The main purpose of the research part is to clarify the need for additional mercury removal in different power plants and define the costs related to the additional removal. Three different mercury-specific technologies are chosen: untreated activated carbon injection, brominated activated carbon injection and bromine addition. The main goal of this study includes various sub-goals. Firstly, the mercury content of coal will be figured out. Secondly, the baseline removal efficiency and baseline mercury emissions across an existing flue gas cleaning configuration will be determined. Thirdly, the requirement of additional mercury removal will be defined. Finally, the costs related to the additional removal by different technologies will be calculated.

LITERATURE PART

2 Mercury

2.1 Characteristics of mercury

Mercury (Hg) is a shiny, metallic element which occurs naturally in the earth's crust throughout the environment. Mercury has a melting point of -39°C and boiling point of 357 °C, thus appearing uniquely in a liquid form at room temperature. It has a density of 13,5 kg/dm^3, which makes it 13 times heavier than water and also slightly heavier than lead. Elemental mercury is relatively insoluble in water (49 μg/dm^3 at 20 °C), but its predominant compounds makes it more soluble by magnitude of 4-6. Mercury has two ionic forms with oxidation states +I and +II which are more usual in water than in the atmosphere. (Bank 2012, Seppänen, Kervinen et al. 2005)

2.2 Appearance and applications

In the nature, mercury appears usually in mineral compounds, rather than in elemental form. Elemental mercury can be found in Earth's crust only in some regions of the world. Most common mercury mineral is cinnabar (HgS). Mercury minerals are usually found in deposits formed by hydrothermal activity, which are typically formed at the edge of convergent tectonic plates. Cinnabar deposits have been mined for centuries to produce mercury, but nowadays the mercury demand is primarily met by other means than mining, such as recycling. Small amounts of mercury can be also found in a wide range of other sediments and rocks, and it appears as an impurity in fossil fuels, especially in coal. (Bank 2012, Kirby, Rucevska et al. 2013)

Elemental mercury has been used in various applications throughout history, such as barometers, thermometers, electrical switches, fluorescent light bulbs, batteries, cosmetic products and dental fillings. Mercury is still commonly used in health care equipment, for example in blood pressure devices and other measuring equipment, but the usage is declining. (Kirby, Rucevska et al. 2013)

2.3 Mercury as a pollutant

Although mercury is a natural element in the environment, it is considered as a global pollutant due to its ability to long-range transport, persistent nature and adverse effect on the human and ecosystem health (Liu, Cai et al. 2011). Since the Industrial Revolution, mercury burden in the environment has been increased significantly by human activity, especially by coal burning, metal ore smelting and gold mining (UNEP 2013). As an indestructible element, mercury accumulates in the environment and cycles between atmosphere, water, and soil. To understand the polluting characteristics of mercury, it is necessary to know the different forms of mercury and their behaviour in the environment.

2.3.1 Speciation

The forms of mercury are also known as mercury speciation. Mercury emissions can be classified in three main forms: gaseous mercury (Hg^0), oxidized mercury (Hg^{2+}) and particulate mercury (Hg(p)). Gaseous mercury, also elemental or atomic mercury, means pure mercury vapour with no chemical bonds. Oxidized mercury means gaseous or solid inorganic mercury compounds, where Hg^{2+} is usually bound with Cl_2 (g), SO_4 (s), O(s,g) or S(s). Mercury can also be adsorbed on to surface of flue gas particulate matter, forming particulate mercury. (Galbreath, Zygarlicke 2000)

Hg^{2+} and Hg(p) are usually easily captured by air pollution control devices of a power plant, which makes it preferable to increase their concentrations in the power plant flue gases. On the contrary, Hg^0 is extremely volatile and insoluble, which makes it chemically more passive and more challenging to be captured in conventional air pollution control devices. This persistent nature of Hg^0 makes it problematic and undesirable form of mercury. While Hg^{2+} and Hg(p) are often deposited locally or regionally, Hg^0 tends to spread further into the atmosphere. (Galbreath, Zygarlicke 2000)

2.3.2 Behaviour in the environment

As an elemental metal, mercury cannot be destroyed, so its amount in the earth remains constant. Once mercury is released into the environment, it cycles through atmosphere, soil and water areas, until it is removed from the cycle by burial in deep ocean sediments or lake sediments, or bound in stable mineral compounds. The process is called mercury cycling. (UNEP 2013)

Mercury enters to the atmosphere mainly as gaseous mercury Hg^0, which is able to spread long distances around the globe. Because of the chemical stability, the residence time of Hg^0 in the atmosphere can be about several months to over one year, and it is estimated that Hg^0 accounts for more than 90% of the total mercury in atmosphere (Zhang, Wang et al. 2016). Rest of the atmospheric mercury is oxidized (Hg^{2+}) or particle bound mercury (Hg(p)), which are more prone to deposit locally and regionally among wet and dry deposition. In water and soil areas, mercury is mostly present as various Hg^{2+} compounds, including inorganic and organic mercury. As a result of sunlight, temperature and biological interactions, Hg^{2+} and Hg(p) can be converted back to Hg^0, which is then re-emitted back to the atmosphere. (Liu, Cai et al. 2011)

Methylmercury is an organic form of mercury, which is formed from inorganic mercury in aquatic environment, such as water areas and wetland soils. It is formed as a result of microbial metabolic processes, where a methyl group is attached to mercury compounds. Methylmercury is in a special concern, because it is highly toxic to humans and wildlife. It has a tendency to bioaccumulate in the environment, which means mercury accumulation into living organisms. Methylmercury is enriched in the food web, which results in high mercury concentrations in some predatory animals. Especially this is the case in large predatory fishes, such as sharks, tuna and whales. (Spaeth, Tsismenakis et al. 2010)

Due to the bioaccumulation of methylmercury, very high mercury concentrations can be found in some wildlife animals. For example, plankton may contain 10 000 times higher

concentrations of methylmercury than surrounding seawater. Mercury enriches upwards in the food web, leading to higher concentrations in predatory animals. Variation of mercury content in animals over the time can be observed by comparing animal tissues from different eras. Studies imply that mercury levels have increased by 12 times in Arctic marine wildlife when compared to pre-industrial stage. Results indicate that 90% of the mercury in these mammals originates from anthropogenic sources. Similar comparison of seabird egg shells in the South China Sea revealed a mercury content increase by magnitude of 10 between the years 1800 and 2000, and an especially rapid increase after 1970. (UNEP 2013)

2.3.3 Toxicology

Mercury is a xenobiotic element, which means it has no physiological role in a living organism. Instead of that, it is highly toxic for humans, wildlife and environment. The toxic nature of mercury is based on mercury's adverse effect on metabolic enzymes within cells. Because of its persistent nature and widespread use, traces of mercury can be found all over the environment, also in human blood, hair and nails. Normally these concentrations are too low to cause harmful effects, but can exceed toxic levels due to bioaccumulation in the food chain. (Spaeth, Tsismenakis et al. 2010)

All forms of mercury are toxic, but methylmercury is considered as the most toxic form because its high absorption rate in the digestive system and the ability to penetrate in tissues. It passes the blood-brain barrier more easily than other forms of mercury, thus having severe effects on brains and central nervous system. As a result of exposure, brain and nervous system may be permanently damaged. Common symptoms for methylmercury intoxication are tremor, convulsion, numbness, headaches, visual impairment, deafness, blindness and death. In addition to neurotoxicity, all forms of mercury may also cause renal toxicity, myocardial infarction, immune malfunction and irregular blood pressure. (Spaeth, Tsismenakis et al. 2010, Liu, Cai et al. 2011)

In general, the developing nervous system of infants and children is more vulnerable to mercury intoxication. Methylmercury is able to permeate into fetus through umbilical cord, which can cause abnormalities to the developing fetal brain. Fetal exposure may lead to sensory and motor defects and other serious neurological disorders of a child. That is why pregnant women or women in fertile age are under special concern for mercury intoxication. (Liu, Cai et al. 2011)

Acute mercury poisoning incidents are nowadays rare, and the main exposure occurs through diet, as the mercury bioaccumulates in seafood, especially in predatory fish (Liu, Cai et al. 2011). Another mercury source may be amalgam fillings, since they have been measured to release small amounts of elemental mercury. However, the toxicity of amalgams remains controversial, because correlation between people with amalgams and symptoms or disorders has not been able to be proved in several scientific studies. (Spaeth, Tsismenakis et al. 2010)

Among wildlife, birds seems to be especially sensitive to mercury. Several field studies on different bird species have pointed out that mercury has particular adverse effects on bird reproduction. In as study made in Maine and New Hampshire, USA, methylmercury was identified to be the main factor responsible for a 40% decline in reproductive success of loons

over an 11-year period. This was mainly caused by decrease in behavioural, physiological and survival activity. (Bank 2012)

2.4 Behaviour of mercury in power plant boilers

In order to understand the mercury emissions and mercury control technologies, it is important to know the behaviour of mercury during combustion process. The flue gas mercury speciation is defined by various complex chemical processes during and after the combustion. These processes are depicted in the Figure 1.

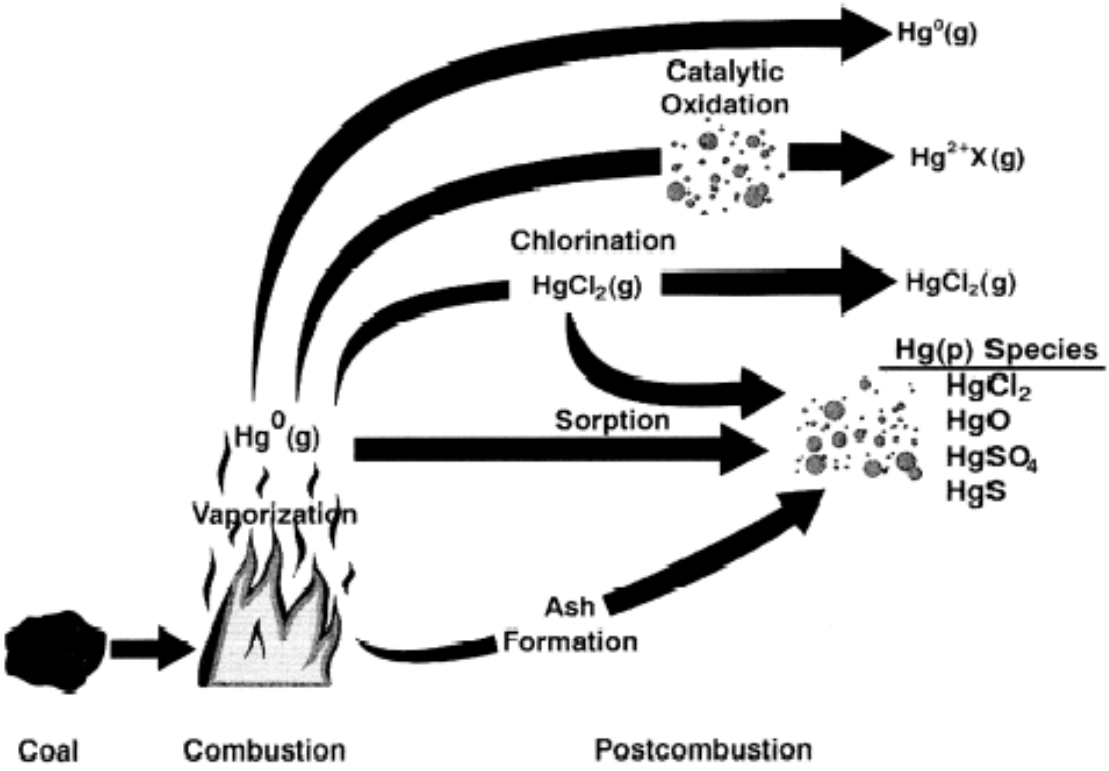

Figure 1: Transformations of mercury in a combustion power plant. (Galbreath, Zygarlicke 2000)

During combustion, mercury is fully vaporized into the flue gas as gaseous mercury (Hg^0). As the temperature of the flue gas decreases, oxidizing and adsorbing reactions start to occur. Inorganic and carbonaceous ash particles play a significant role in these processes. Particles act as catalysts for the oxidation reactions between mercury and other chemical species in the flue gas, which results in formation of Hg^{2+} -compounds. Fly ash particles also promote the mercury adsorption, which leads to formation of Hg(p). The adsorption can occur via physical adsorption, chemisorption, chemical reactions or a combination of these. (Galbreath, Zygarlicke 2000)

According to several researches and field observations, chlorine is a very important chemical in mercury oxidation. This emphasizes the importance of initial chlorine concentration of the fuel in regard to the final mercury speciation. In the mercury chlorination process, Hg^0 (g) reacts

with hydrogen chloride (HCl (g)) or chlorine gas (Cl_2 (g)) and forms mercury chloride ($HgCl_2$ (g)). Theoretical studies and investigations suggest that Cl_2 is much more reactive chlorinating agent than HCl. Chlorine is released in the combustion process primarily as HCl, and some of the HCl is then oxidized into Cl_2 and H_2O, following the equation 1. However, the reaction requires a catalyst and the concentrations of formed Cl_2 are usually low (1%). Despite the concentration is low, it is still likely to be excessive in relation to Hg^0 content. (Galbreath, Zygarlicke 2000)

$$4HCl\ (g) + O_2\ (g) \rightarrow Cl_2\ (g) + H_2O\ (g) \quad (1)$$

Other flue gas components, such as O_2, CO, SO_2, SO_3, NO and NO_2, have also influence in the mercury speciation. The influence of some flue gas components can be found in the Table 1. Oxygen (O_2 (g)) seems to promote the adsorption of mercury on the fly ash particles, especially in temperatures between 100 °C and 300 °C, which increases the amount of Hg(p). Oxygen can also oxidize mercury into mercury oxide (HgO (g)). However, HgO (g) may be reduced to Hg^0 (g) by chemical reactions with SO_2 (g) or CO (g). Also nitrogen oxide (NO (g)) seems to inhibit Hg^{2+} formation. By contrast, nitrogen dioxide (NO_2 (g)) promotes formation of Hg^{2+} but inhibits formation of Hg(p). (Galbreath, Zygarlicke 2000, Zhang, Wang et al. 2016)

Besides of chlorine, the sulphur content of the fuel has also much influence in the mercury speciation. Presence of sulphur oxide (SO_2 (g)) seems to inhibit the formation of Hg^{2+} by reacting with Cl_2, forming HCl and sulphur trioxide (SO_3 (g)). This reaction reduces the amount of Cl_2, which is an important mercury oxidizer. The reaction between chlorine and sulphur dioxide is following:

$$Cl_2\ (g) + SO_2\ (g) + H_2O\ (g) \rightarrow 2HCl(g) + SO_3\ (g) \quad (2)$$

This is why high sulphur-chlorine ratio of fuel can result in poor mercury oxidation in the flue gas. On the other hand, SO_2 may promote mercury adsorption on the fly ash particles via series of chemical reactions. A part (1-3%) of the released SO_2 is usually naturally oxidized to SO_3. SO_3 reacts water and forms gaseous sulphuric acid (H_2SO_4 (g)), which may condensate on ash particle surfaces as a liquid H_2SO_4 (l). The liquid H_2SO_4 (l) is able to promote mercury adsorption through series of chemisorption processes, which result in the formation of mercury sulphide HgS (s, g) on the fly ash particles. (Galbreath, Zygarlicke 2000)

Table 1: Influence of some flue gas chemicals on the mercury speciation.

Chemical	Hg^0	Hg^{2+}	Hg(p)
HCl (g), Cl_2 (g)	decreasing	increasing	increasing
O_2 (g)	decreasing	increasing	increasing
CO (g)	increasing	decreasing	decreasing
N_2 (g)	decreasing	increasing	increasing
SO_2 (g)	increasing	decreasing	decreasing
SO_3 (g)	decreasing	increasing	-
NO (g)*)	decreasing	increasing	increasing
NO_2 (g)	-	increasing	decreasing
Fe (s)	increasing/decreasing	increasing/decreasing	-
Al (s)	decreasing	increasing	increasing
Sources: (Galbreath, Zygarlicke 2000), *) (Zhang, Wang et al. 2016)			

Zhang et al. have collected data from 30 on-site measurements in coal-fired power plants and industrial boilers. The average proportions of mercury species in the flue gas of these pulverized-coal boilers were 56 % for Hg^0, 34% for Hg^{2+} and 10% for Hg(p). However, the proportions of Hg^{2+} and Hg(p) varied between 5 - 82% and 1 - 28%, respectively. Besides of coal properties, the boiler type was also noticed to affect the mercury speciation. In circulating fluidized bed (CFB) boiler, where more contact between mercury and fly ash is able to occur, the amount of Hg(p) can be as high as 65%. (Zhang, Wang et al. 2016)

In summary, the flue gas mercury speciation after the boiler depends on a series of chemical and physical reactions. The outcome of these reactions depends on fuel composition and combustion environment. The most important substances in the fuel are chlorine, mercury, sulphur and ash. In general, high ash and unburned carbon content in the flue gas increase the amount of Hg(p). Chlorine promotes the formation of Hg^{2+}, while high sulphur content of the fuel results in high levels of SO_2, which in turn reacts with chlorine and reduces the amount of Hg^{2+}.

3 Mercury content of fuels

3.1 Coal

Coal is a diverse fuel as its chemical composition varies a lot depending on the deposit location and age. Furthermore, mining and storage processes alter the chemical composition, though the main characteristics depend on the coal origin. Consequently, also the mercury content shows significant variation. As with any other fuel, total chemical composition of coal is a determining factor for the mercury speciation in flue gas, and eventually for mercury air emissions from a power plant. For better understanding of mercury and other chemical content of coal, it is necessary to clarify different coal types.

Coal is formed from organic matter as a result of hundreds of millions of years long geological processes. Coal formation requires anaerobic environment and high pressure, under which the organic matter does not decay, but moisture and gases are diminished. The organic matter undergoes several processes, during which its composition is reformed. The chemical composition of the coal depends on the initial chemical composition of its parent plant, nature and extent of the changes, and the nature of inorganic matter that has been present during the formation. (Alakangas 2000, European Commission 2016)

There are couple of different ways to classify different coal types. According to the ASTM classification that has been developed in the US, there are four classes of coal: anthracite, bituminous, subbituminous and lignite. The classification is done based on the carbon and volatile matter content and the calorific value. (European Commission 2016)

According to the UN/ECE international classification system, there are higher rank coals and lower rank coals, based on the gross calorific value and vitrinite mean random reflectance in oil. Lower rank coals can be further divided into sub-bituminous coals and lignite. Anthracite and bituminous coal are seen as higher rank coals. (European Commission 2016)

The age of different coal types is the following from the youngest to the oldest one: lignite, subbituminous coal, bituminous coal and anthracite (Alakangas 2000). Older coal types have higher carbon content and lower water content, and also higher energy content (kJ/kg) than the younger variants. Thus, anthracite and bituminous coals are referred as high rank coals. Consequently, lignite and subbituminous coal, which have poorer energy content, are referred as low rank coals. (Center for Climate and Energy Solutions 2016). Furthermore, in some contexts definitions hard coal and brown coal are used. Hard coal usually means higher rank coals, such as bituminous coal, and brown coal means lower rank coals, such as subbituminous and lignite coal.

The properties of coal types according to the ASTM classification are represented in the Table 2. The data is collected for coals burned in U.S. Power Plants. The variation of chemical composition of different coal types can be clearly seen in this Table 2. Especially the chlorine content is especially interesting, because chlorine is a determining element in mercury oxidation. It can be seen that bituminous coal contains usually more chlorine than lower rank coals. As mentioned in chapter 2.4, this results in higher formation of oxidized mercury in

combustion process. Nevertheless, the mercury and sulphur contents of subbituminous are usually lower, resulting also in lower emissions of these chemicals. High sulphur may inhibit the oxidizing effect of chlorine, which results in higher mercury emissions. Also the calorific value of different coal types has to be taken into account when estimating the emissions per produced electricity unit. (Strivastava 2010)

Table 2: Properties of different coal types (ppm = parts per million = mg/kg).

Coal type	Anthracite	Bituminous coal	Subbituminous coal	Lignite
Net calorific value (MJ/kg) (dry)	Average: 34,9[2)]	Average: 30,7[1)] Range: 20,1 – 32,6[1)]	Average: 27,9[1)] Range: 20,0 – 30,6[1)]	Average: 23,3[1)] Range: 22,1 – 24,9[1)]
Carbon (%) (dry)	Range: 86 – 97[2)]	Range: 45 – 86[2)]	Range: 35 – 45[2)]	Range: 25 – 35[2)]
Mercury (ppm) (dry)	Range: 0,06-0,2[3)]	Average: 0,113[1)] Range: 0,036 – 0,279[1)]	Average: 0,071[1)] Range: 0,025 – 0,136[1)]	Average: 0,107[1)] Range: 0,080 – 0,127[1)]
Chlorine (ppm) (dry)	-	Average: 1033[1)] Range: 48 – 2730[1)]	Average: 158[1)] Range: 51 – 1143[1)]	Average: 188[1)] Range: 133-233[1)]
Sulphur (%) (dry)	Range: 0,5 – 0,8[3)]	Average: 1,69[1)] Range: 0,55 – 4,10[1)]	Average: 0,50[1)] Range: 0,22 – 1,16[1)]	Average: 1,30[1)] Range: 0,8 – 1,42[1)]
Ash (%) (dry)	-	Average: 11,1[1)] Range: 5,4 – 27,3[1)]	Average: 8,0[1)] Range: 4,7 – 26,7[1)]	Average: 19,4[1)] Range: 12,2 – 24,6[1)]
Sources: 1) (Strivastava 2010), 2) (Center for Climate and Energy Solutions 2016) 3) (European Commission 2016)				

According a research note made by Technical Research Centre of Finland (VTT), which has investigated the characteristics of fuels, the mercury content of coal used in Finland is 0,11 mg/kg (dry) on average. The variation is between 0,03-0,23mg/kg. (Alakangas 2000). According to another research note by VTT, the mercury content of coal burned in Finland is 0,09 mg/kg on average, varying between 0,005-0,15 mg/kg (Hepola 2003).

In another report made by Lyyränen, Ohlström et. al. (2014), information on mercury content in coal has been collected from several other literature sources. The results are presented in the Table 3. The values from sources Taipale (1996) and Tolvanen (2004) regarding the coal used in Finland are quite similar to those than reported by Alakangas (2000) and Hepola (2003). The average mercury content of all the values represented in the Table 3 is 0,35 mg/kg, while the range varies from 0,01mg/kg to 1mg/kg. According to this, the variation is very high, as the highest value can be 100 times higher than the lowest value.

Table 3: Mercury content of coal as reported in different sources. (Lyyränen, Ohlström et al. 2004)

Reference	Davidson, 2000	Sloss, et al. 2000	Clarke and Sloss, 1992	Taipale, 1996 (coal used in Finland)	Tolvanen, 2004 (coal used in Finland)	All references
Average (mg/kg)	0,09	0,11	0,5	0,11	0,13	0,35
Range of variation (mg/kg)	0,01 - 0,18	-	0,02 - 1	0,03 - 0,23	0,07 - 0,21	0,01 - 1

In Poland, a country with a lot of coal-fired power plants, several studies have been made regarding the mercury content of coal. According to a study made by Lorenz, Grudziński (2008), the average mercury content of Polish coal is 0,1 - 0,15 mg/kg for hard coal and 0,3 - 0,35 mg/kg for brown coal. However the variation was very high, 0,01 - 0,967 mg/kg for hard coal and 0,08 - 1,03 mg/kg for brown coal. The content of mercury in brown coal is 2 - 4 times higher than in hard coal. Because of the lower calorific value of brown coal, the mercury content related to amount of energy produced is about 5 times higher in brown coal than hard coal. (Lorenz, Grudziński 2008)

Pirrone, Cinnirella et al. (2010) have collected data from different sources in order to estimate the mercury content of coal in different geographical locations. The data is presented in the Table 4. In some cases, also mercury content of fly ash is reported. The Table 4 shows clearly the high variation of mercury content, as the value ranges from 0,01 mg/kg to 1,95 mg/kg. It is notable, that the highest variation counts to European coal. According to the Table 4, relatively low mercury contents can be found in Korea, Japan, Colombia, Venezuela and USA. Anyway, it has to be remembered that mercury content of coal can show significant local variation, so no nationwide generalizations can be made from the data. (Pirrone, Cinnirella et al. 2010)

Table 4: Mercury content of coal in different geographical regions. (Pirrone, Cinnirella et al. 2010)

Country/region	Hg in coal (mg/kg)	Hg in coal fly ash (mg/kg)
Australia	0.01–1.0	0.34
China	0.19–1.95	–
Guizhou Province	0.52	–
Europe	0.01–1.5	0.23
India	0.11–0.80	0.007-0.28
Japan	0.045	-
Korea	0.012–0.048	–
Russia	0.02–0.9	
South Africa	0.01–1.0	0.56–0.64
Argentina	0.021–0.96	
Brazil	0.041–0.778	
Colombia	0.020–0.17	
Peru	0.041–0.63	
Venezuela	0.030–0.280	0.268
USA	0.17 (mean)	
World	0.02–1.0	0.62

As a conclusion, it has to be mentioned that there is significant variation in the chemical composition of coal. The content of different trace elements, also mercury, depends much on the coal origin, and significant variation may also occur within the same region. Thus, the mercury content of coal burned in a power plant is very site-specific, and should by analysed case by case in order to acquire reliable information. Some conclusions about the mercury content can be made by coal type. Although the concentrations of mercury in coal are usually relatively low, the high volume of coal burned globally leads to significant emissions in the air.

3.2 Peat

Peat constitutes of residues of plants, which have not decayed completely. As mentioned before, peat is organic matter, which is buried under anaerobic conditions. As a result of high water content and absence of oxygen, the organic matter does not decay completely and starts to accumulate slowly. Peat is usually found in swamps and wetlands. A major fraction of peat consists of coal. The coal share varies depending on the peat type and the time of decay, but a typical value is 53-56%. Rest of the peat comprises mostly oxygen, but also small fractions of hydrogen, sulphur and nitrogen. The heating value of dry peat is typically 20,9 - 22,5 MJ/kg. (Alakangas 2000)

According to a research note of VTT, mercury content of Finnish peat varies between 0,06 - 0,075 mg/kg, being 0,07 mg/kg on average. This result suggests that the mercury content of Finnish peat is averagely lower than the mercury content of coal used in Finnish power plants. (Pohjola, Hahkala et al. 1983). Another VTT research note says that the mercury content in the ash residual from peat is approximately 0,3-2mg/kg (dry). (Alakangas 2000)

A Finnish state-owned energy company Vapo has done a research on heavy metal contents in Finnish peat production zones. In total, 46 samples from different depths (in range 0-200cm) were taken in three different zones in the Central Finland. The average mercury content of these samples was 0,038 mg/kg, while the range was between 0,005 - 0,08 mg/kg. There was no clear correlation between the depth and mercury content. As a reference, the research report also included Vapo's internal data of mercury content in produced and delivered peat between the years 1990 – 2013. The average mercury content of 64 samples was 0,057 mg/kg, and the range was 0,02 – 0,18 mg/kg. (Lehtovaara, Ovaskainen et al. 2014)

The same report cites also several other studies on peat mercury content in different regions. In studies made in Sweden and Norway, samples were taken from different depths in natural peatland areas, also swamps. The results indicated that mercury content in natural peatlands is clearly highest in depths of 10-30cm. The Norwegian study gave an average mercury content of 0,098mg/kg (range 0,047 – 0,139 mg/kg) for 30cm deep surface layer. In the Swedish study, an average mercury content of 0,108 mg/kg (range 0,053 – 0,179 mg/kg) was measured for a layer of same depth. In the both studies, the mercury content decreased and stabilized after the surface layer and was averagely 0,015 – 0,020 mg/kg in the deeper levels. A study made on heavy metal contents in Estonian peat reported an average mercury content of 0,050 mg/kg (range 0,003 – 0,066 mg/kg). (Lehtovaara, Ovaskainen et al. 2014)

It seems that the mercury content in Finnish peat is similar to contents measured in Norway, Sweden and Estonia. Natural peatlands seems to be richer in mercury near to surface, while in areas that are dedicated for energy peat production, the values are independent from depth. The mercury content in Finnish peat production areas is lower than the mercury content in surface levels of Norwegian and Swedish natural peatlands. However, in deeper levels, the Finnish mercury content seems to be higher than the mercury content in natural peatlands in corresponding depth. This indicates that the mercury content of peat production areas has levelled off due to mixing of different layers. (Lehtovaara, Ovaskainen et al. 2014). Higher mercury content of the surface levels in natural peatlands may indicate the increased air emission fallout during the industrialization.

3.3 Crude oil

Similar to coal, crude oil is formed from the organic matter residuals during millions of years. Crude oil is composed of hydrocarbons of different lengths. The smallest hydrocarbons are gaseous, as the heaviest ones have a boiling point as high as 700 °C. (Alakangas 2000)

Mercury content of oil is generally low, however the values varies highly according to the crude oil type. Usually the values are lower than in coal. According to Pirrone, Cinnirella et. al. (2010), the mercury content ranges from 0,007 to 30mg/kg, and a typical value is 3,5 mg/kg. Mercury content also depends on the fraction size. Heavy oil refinery fractions include more mercury than lighter fractions. VTT research note reports that mercury content of oil is usually less than 0,3 mg/kg (Alakangas 2000). Another VTT research note states that mercury content of crude oil is generally low, and 0,01 mg/kg is an usual value. However occasionally the content can be as high as 30mg/kg (Hepola 2003). These values are consistent to that reported by Pirrone Cinnirella et. al. (2010).

3.4 Summary

The mercury content of coal, peat and oil are summarized in the Table 5, which contains data from different sources. It seems that mercury content in these fuels varies significantly depending on the deposit. Mercury content of coal varies a lot according to the location and coal rank. Lignite and other lower rank coals seem to contain more mercury than hard coal. The average mercury content in peat seems to be little lower than in coal, and the variation seems to smaller. However, data for this study is gathered only from Finland, Sweden, Norway and Estonia, which constitutes a relatively small geographical area. In natural peatlands, mercury content tends to be higher in surface levels. Mercury content in crude oil can be high, and according to some sources, the average value can be much higher than in coal. However, it has to be mentioned that mercury content of oil is reduced during the refining process, and the final petroleum products do not usually contain much mercury, if any.

Table 5: Summary of mercury content in coal, peat and oil.

Fuel	Average (mg/kg)	Variation (mg/kg)	Country of origin	Remarks
Coal	0,11[1)]	0,03 – 0,23[1)]	-	Coal used in Finland
	0,09[2)]	0,005 – 0,15[2)]	-	Coal used in Finland
	0,35[3)]	0,001 – 1[3)]	-	Data gathered from different literature sources
- Hard coal	0,1 – 0,15[4)]	0,01-0,967[4)]	Poland	Coal from Polish mines
- Brown coal	0,3 – 0,35[4)]	0,08-1,034[4)]	Poland	Coal from Polish mines
Peat	0,07[2)]	0,06 – 0,075[2)]	Finland	
	0,038[7)]	0,005-0,08[7)]	Finland	Peat production zone
	0,057[7)]	0,02 – 0,018[7)]	Finland	Peat production zone
	0,098[7)]	0,047 – 0,139[7)]	Norway	Natural swamp, measured in 0-30 cm from the surface level
	0,108[7)]	0,053 – 0,179[7)]	Sweden	Natural swamp, measured in 0-30 cm from the surface level
	0,050[7)]	0,003 – 0,066[7)]	Estonia	Peat production zone
Oil	3,5[5)]	0,007 – 30[5)]	-	
	<0,3[1)]		-	
	0,01[6)]		-	
Sources: 1) (Alakangas 2000), 2) (Pohjola, Hahkala et al. 1983), 3) (Lyyränen, Ohlström et al. 2004), 4) (Lorenz, Grudziński 2008) 5)(Pirrone, Cinnirella et al. 2010) 6) (Hepola 2003) 7) (Lehtovaara, Ovaskainen et al. 2014)				

4 Mercury emissions

Mercury is emitted into the atmosphere, water and soil by natural events and human action. Emission sources are usually divided into natural sources and anthropogenic sources. Natural sources include the primary natural sources and re-emissions of already deposited mercury from land and water areas. Primary natural sources consist of geological processes, such as volcano eruptions, other geothermal activities and emissions from mercury enriched topsoil. Re-emissions include the release of mercury that is already deposited onto soils, waters and vegetation from the past emissions. Anthropogenic emissions comprise emissions from human activity, mostly energy and industrial sector. (Pirrone, Cinnirella et al. 2010). It is noteworthy that although re-emissions are often classified as a natural source, it is impossible to determine the exact origin of mercury. Thus re-emissions may also include mercury that is originally released by anthropogenic sources.

Mercury emissions are mostly based on estimations, because accurate data is rarely available. Mercury emission assessments require a lot of assumptions and generalizations, and uncertainty arises from the correctness of the data, the validity emission factors applied and the effectiveness and use of emission control technologies. Measurements are often based on only a few measurement points and short measuring period, and the data is then extrapolated to produce annual emission data. Furthermore, national emissions reports use different reporting methods, which complicates the comparison of data. Due to uncertainties in mercury emission estimations, accurate reliable values cannot be given. Instead, a range of values is preferred. (UNEP 2013)

4.1 Total emissions

According to the Global Mercury Assessment report 2013 by UNEP, yearly global mercury emissions are estimated to be between 5500 and 8900 tons. The UNEP reports provide information about mercury emissions in intervals of five years, and the data of the newest report is for the year 2010. According to that, anthropogenic emissions contribute about 30% of the global emissions, while 70 % are from natural sources. (UNEP 2013)

Liu, Cai et al. have also introduced estimations that are collected from other literature sources. According to them, global mercury emissions are 5000 – 6000 tons/year. Natural emissions are in the range of 1800 – 3000 tons/year and anthropogenic emissions in the range of 2000 – 2600 tons/year. (Liu, Cai et al. 2011)

4.2 Natural sources

Natural emissions comprise 70% of all mercury emissions. Mercury is emitted to the air from soils and water areas due to natural processes that convert inorganic mercury into volatile gaseous mercury. Mercury may also be re-emitted during forest fires and biomass burning. Estimating the amount of re-emissions is difficult and is often done by different models, which are based on data on mercury chemistry in water, land and atmosphere. The share of re-emissions is estimated to be 60% of all mercury emissions. (UNEP 2013)

Oceans are by far the most important natural source of mercury, representing 52% of all natural emissions (primary emissions + re-emissions), followed by biomass burning with a share of 13 %. The high share of oceans emissions results from the large area of oceans. In relation to surface area, land areas emit more emissions than water areas. Volcanoes contribute 2% of the natural emissions. (Pirrone, Cinnirella et al. 2010)

4.3 Anthropogenic sources

Anthropogenic mercury emissions sources are mostly point sources, such as combustion plants and industrial installations. Combustion sources include fossil-fuel fired power plants, waste incinerators, municipal waste combustors and sewage sludge incinerators. Coal burning is one of the most significant mercury emission sources to the atmosphere. Industrial installations comprehend especially metal and cement production facilities and chlor-alkali industry. They release mercury both in air and water. Chlor-alkali industry uses mercury-cell technology in chlorine and soda production, but the trend is decreasing. Also, artisanal and small-scale gold mining are a significant emission source, especially in South-America. (Liu, Cai et al. 2011)

The estimation of Global Mercury Assessment report for global anthropogenic mercury emissions is 1960 tons in the year 2010, with a variation of 1010 – 4070 tons/year. Artisanal and small-scale gold mining (410 – 1040 tons/year) and coal burning (304 – 678 tons/year) are the most significant anthropogenic emissions sources. Other remarkable sources are production of ferrous and non-ferrous metals, and cement production. According to the newest estimations, the emissions from artisanal and small scale-gold mining are 727 tons, meaning 37% of total anthropogenic emissions. Values for artisanal and small-scale gold mining are very hard to calculate, because the sector is dispersed, uncontrolled and may also be illegal. This is why estimations contain significant uncertainties. This is followed by coal burning, responsible for 474 tons (24% of total anthropogenic emissions). Other fossil fuels, such as oil and natural gas are low, contributing only 10 tons of annual mercury emissions. This is because mercury is removed from crude oil and natural gas prior to combustion, and disposed of in landfills. Mercury emissions from different sources are shown in the Table 6. (UNEP 2013)

Table 6: Anthropogenic mercury emissions from different sectors. (UNEP 2013)

Sector	Emission (range), tonnes	%
By-product or unintentional emissions		
Fossil fuel burning		
Coal burning (all uses)	474 (304 - 678)	24
Oil and natural gas burning	9.9 (4.5 – 16.3)	1
Mining, smelting, & production of metals		
Primary production of ferrous metals	45.5 (20.5 – 241)	2
Primary production of non-ferrous metals (Al, Cu, Pb, Zn)	193 (82 – 660)	10
Large-scale gold production	97.3 (0.7 – 247)	5
Mine production of mercury	11.7 (6.9 – 17.8)	<1
Cement production	173 (65.5 - 646)	9
Oil refining	16 (7.3 - 26.4)	1
Contaminated sites	82.5 (70 - 95)	4
Intentional uses		
Artisanal and small-scale gold mining	727 (410 – 1040)	37
Chlor-alkali industry	28.4 (10.2 – 54.7)	1
Consumer product waste	95.6 (23.7 – 330)	5
Cremation (dental amalgam)	3.6 (0.9 - 11.9)	<1
Grand Total	**1960 (1010 – 4070)**	**100**

4.4 Mercury emissions in the world

The global distribution of emission sources is shown in the Table 7. According to data collected for the year 2010, about 40% of the global anthropogenic mercury emissions are originating from Asia, most of them from East and Southeast Asia. East Asia is the major regional source for long-range air emissions. China alone contributes about 75 % of the emissions in this region, which means about one third of the total global emissions, mainly due to increasing coal combustion for power production. (UNEP 2013)

The estimation of the worldwide mercury emission trend in the past is challenging due to differences in methods used to produce inventory data, and the sectors included to the inventory

have not been the same over time. The worldwide mercury emissions have likely peaked in the 1950s to 1970s and decreased after that. The main decrease has occurred in Europe, Russia and North America due to increased implementation of air pollution control systems. On the other hand, mercury emissions in Asia have been increasing, and the total worldwide emissions maybe increasing again. (UNEP 2013)

Table 7: Global distribution of mercury emissions in 2010 and their shares of the total anthropogenic output. (UNEP 2013)

Region	Emission (range), tonnes	%
Australia, New Zealand & Oceania	22.3 (5.4 - 52.7)	1.1
Central America and the Caribbean	47.2 (19.7 - 97.4)	2.4
CIS & other European countries	115 (42.6 - 289)	5.9
East and Southeast Asia	777 (395 - 1690)	39.7
European Union (EU27)	87.5 (44.5 - 226)	4.5
Middle Eastern States	37.0 (16.1 - 106)	1.9
North Africa	13.6 (4.8 - 41.2)	0.7
North America	60.7 (34.3 - 139)	3.1
South America	245 (128 - 465)	12.5
South Asia	154 (78.2 - 358)	7.9
Sub-Saharan Africa	316 (168 - 514)	16.1
Undefined (global total for emissions from contaminated sites)	82.5 (70.0 - 95.0)	4.2
Grand Total	**1960 (1010 – 4070)**	**100**

4.5 Mercury emissions in Europe

The European Pollutant Release and Transfer Register (E-PRTR) provides environmental data from more than 30 000 emission sources across the Europe. The register includes EU Member States, Iceland, Liechtenstein, Norway, Serbia and Switzerland. Based on the data of E-PRTR, mercury emissions in Europe were 29,5 tons in 2013, from which 25,1 tons were air emissions. The highest contributor is energy sector with air emissions of 14,2 tons/year, which is 57% of all air emissions. Energy sector emissions are mostly from combustion power plants. Other major air emission sources are metal production (4,4 tons), mineral industry, mostly cement production (2,9 tons), and chemical industry (2,4 tons). The emission sources by sector are represented in the Figure 2. (European Environment Agency 2016)

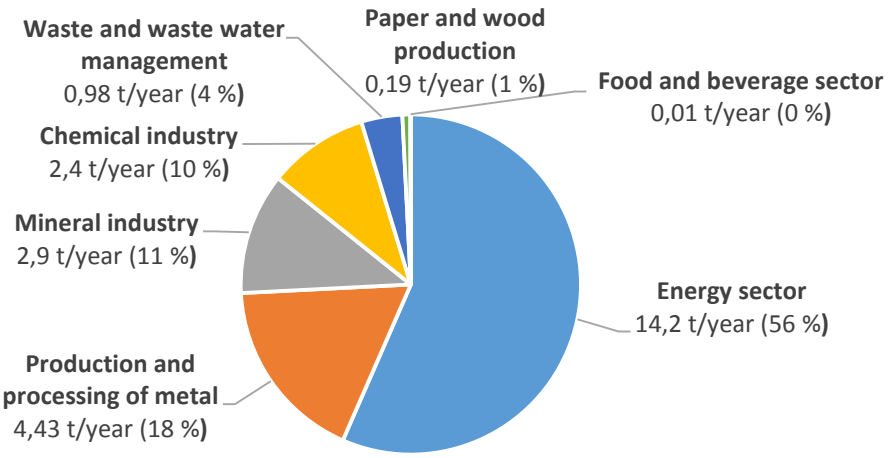

Figure 2: Mercury air emissions by sector in Europe in 2013. Data source: E-PRTR. (European Environment Agency 2016)

The major point sources in Europe are coal-fired power plants, mostly in Germany. Mercury air emission distribution within European countries is shown in the Figure 3. According to the data, Germany releases by far most air emissions with a contribution of 6,94 tons and 28% share. This is followed by Poland with 3,32 tons (13%) and United Kingdom with 2,85 tons (11%). Together, these three countries are responsible for more than half of all air emissions in Europe. (European Environment Agency 2016)

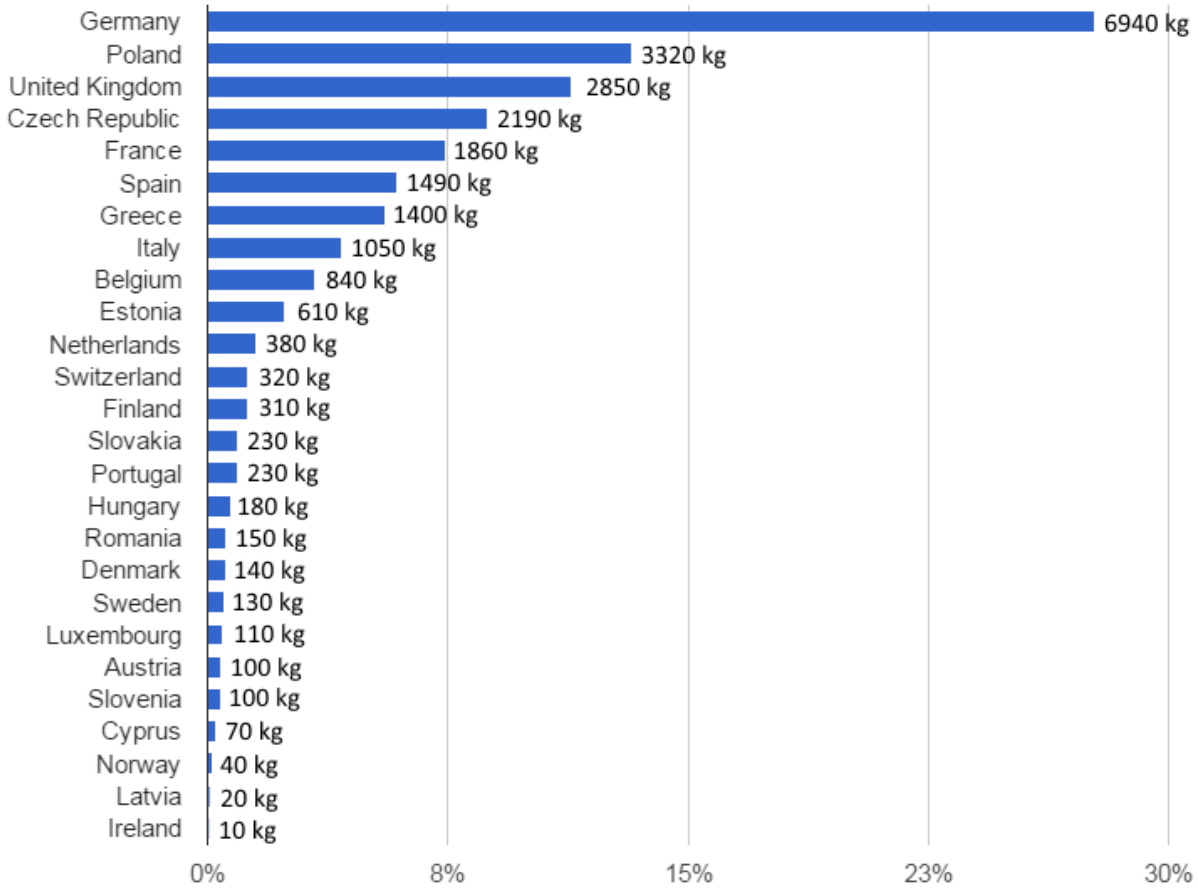

Figure 3: Mercury air emission distribution in European countries in 2013. Data source: E-PRTR. (European Environment Agency 2016)

4.6 Mercury emissions in Poland

Most of the anthropogenic mercury emissions in Poland originate from coal combustion. Polish energy production is heavily dependent on coal, as about 90% of electricity and heat are produced by coal (data from the year 2011) (World Energy Council 2016). This ranks Poland as the second largest coal consumer in the EU, right after Germany. Thus, Poland is also one of the largest anthropogenic mercury emitters in Europe. Although coal consumption and mercury emissions have been decreasing, it is estimated that Poland will remain one of the highest emitters in Europe. Mercury emissions in Poland can be reduced by substituting coal by other energy sources, but this will likely not happen in the near future. Another option is to reduce mercury emissions by efficient air pollution control devices. (Glodek, Pacyna 2009)

According to the data in the E-PRTR, total mercury emissions in Poland were 3,98 tons in 2013, from which 3,32 tons were air emissions. The major air emission source is energy sector. Thermal power plants contribute 2,59 tons of air emissions, which covers 78 % of all emissions to air. Another large emission sources are cement production with 0,4 tons (12% of air emissions) and landfills with 0,23 tons (7% of air emissions). (European Environment Agency 2016)

5 Mercury emission regulation

5.1 International mercury emission regulation

Global mercury emissions are regulated internationally by the Protocol on Heavy Metals, which is one of the eight protocols included in the UNECE Convention on Long-Range Transboundary Air Pollution (LRTAP). UNECE stands for the United Nations Economic Commission for Europe, which is a regional commission of United Nations. LRTAP was entered into force in 1983 and has been one of the main international instruments for the air emission control. (UNECE 2016a)

The Protocol on Heavy Metals was adopted in 2003 and revised in 2012, and it aims to limit air emissions of mercury, lead and cadmium from industrial sources, combustion processes and waste incineration. The Protocol on Heavy Metals sets stringent limit values for emissions from stationary sources and suggests implementation of best available techniques (BAT) in order to meet these values. One of the main obligations for the Parties is to reduce emissions of mercury, lead and cadmium compared to the reference year of 1990 (or an alternative year between 1985 and 1995). (UNECE 2016b)

Building on the Protocol on Heavy Metals, Minamata Convention on Mercury was agreed in 2013 at the fifth session of the Intergovernmental Negotiating Committee on mercury, and was adopted later in the same year. I has been signed by 128 countries, including Finland. Minamata Convention is an international treaty which aims to protect human health and the environment from adverse effects of mercury. The convention was a result of years of negotiations, as it was seen that voluntary actions to restrict mercury emissions were not sufficient. The name of the Convention refers to the Japanese city of Minamata, where a serious mercury poisoning incident was discovered in 1956. (UNEP 2016)

The main objectives of the Convention are to ban new mercury-mines and phase out the existing ones, limit the use of mercury in products and processes, restrict emissions to air, land and water, and control mercury usage in small-scale gold mining. According to the article 8, which states the obligations in regard to mercury emissions into atmosphere, each party of the convention shall take measures to restrict mercury emissions and make a national plan for that purpose. Best available techniques to control the emissions shall be used and implemented not later than five years after the validation of the Convention. (United Nations Environment Programme 2013)

In the USA, the main instrument for mercury control is the Mercury and Air Toxics Standard (MATS). MATS is a part of the Clean Air Act, which is a federal law for air pollution control. Issued in 2011, MATS superseded the former Clean Air Mercury Rule (CAMR), which was based on a cap-and-trade system. MATS sets emission standards for toxic air pollutants, such as mercury, arsenic and other heavy metals. It includes nationwide mercury emission limits on coal-fired power plants, aiming to cut 91% of the mercury emissions from new and existing coal and oil-fired power plants. (US EPA 2016)

5.2 Mercury emission regulation in Europe

In the European Union, emissions are mainly controlled by EU directives. Large Combustion Plant Directive (LCP, Directive 2001/80/EC) was the main directive for setting limit values for flue gas emissions in large installations, such as large power plants. It covered the combustion plants with equal or greater than 50 MW of thermal input. The LCP directive entered into force in 2001, and obliged existing plants to comply with the emission limit values (ELVs) by 2008. The aim of the LCP directive was to reduce the emissions of acidifying pollutants, particles and ozone precursors, thus setting ELVs for SO_2, NOx and particle emissions. It was later substituted by the Industrial Emissions Directive. (European Commission 2016c)

Nowadays, the Industrial Emission Directive (IED, Directive 2010/75/EU) is the main pollution control instrument for industrial installations. It is applied to combustion plants with thermal input equal to or greater than 50 MW. The IED entered into force on 6 January 2011 and had to be transposed to national legislation by 7 January 2013. It is based on 7 previously existing directives, substituting also the LCP directive in the beginning of 2016. The IED aims to protect human health and environment from industrial pollutants, in particular by applying the best available techniques (BAT). According to the IED, BAT conclusions shall work as a reference when setting the permit conditions to installations covered by the IED. (European Commission 2016b)

Best available techniques are seen as the most viable techniques to reduce emissions. BATs are defined by experts from the Member states, industry and environmental organizations. The work is coordinated by European IPPC Bureau, which is located in Seville Spain. The process is also known as Seville process. The results of this process are BAT reference documents (BREFs), which will introduce the BAT conclusions. These conclusions are then adopted by the European Commission. (European Commission 2016b)

An important part of the IED are legally binding BREF documents. The emissions levels that must be obliged are introduced in BAT conclusions of BREF documents. BAT conclusions introduce the best available techniques and the emissions levels associated with the technologies, known as emissions levels associated with the best available techniques (BAT-AELs). It contains also information about the accessibility of the techniques and monitoring methods applied for the emissions. (European Commission 2016d)

The BREF documents are meant to be updated periodically. The current BREF version originates from the year 2006, and it has been under a revision process. The final draft of the new BREF version was published in June 2016, and it was confirmed in October 2016. The new BREF will possibly come into force in the beginning of 2018, after which there will be 4 years transition period for parties to adopt the new emission limits. This means the new emission limits have to be achieved by 2022. The timeline of the emission regulation process is illustrated in the Figure 4. (Ekman 2016)

The new BREF will tighten the existing emission levels for nitrogen oxides, sulphur dioxide and particulate matter, and emission levels will be set on new pollutant components, such hydrogen chloride, hydrogen fluoride, carbon monoxide, ammonia, nitrogen dioxide and mercury. Setting a specific emission level also for mercury, the power plant operators will have to consider additional measures to reduce mercury emissions in coal fired power plants. The

emission levels are introduced as a range, which sets upper and lower boundary for the emissions, and a national or local authority is entitled to choose the specific emission level within this range. (Ekman 2016)

The BAT-AELs for mercury included in the new BREF document are shown in the Table 8. The emission values are expressed as micrograms per normal cubic meter (µg/Nm^3). Normal cubic meter is the volume of dry flue gas in standard temperature and pressure, which are 0 °C and 101,3 kPa. The emission levels are determined for existing and new power plants with different fuels and thermal power capacities. Peak- and emergency-load power plants as well as CHP (combined heat and power) plants are excluded from the revised emission limits. (European IPPC Bureau 2016, Ekman 2016)

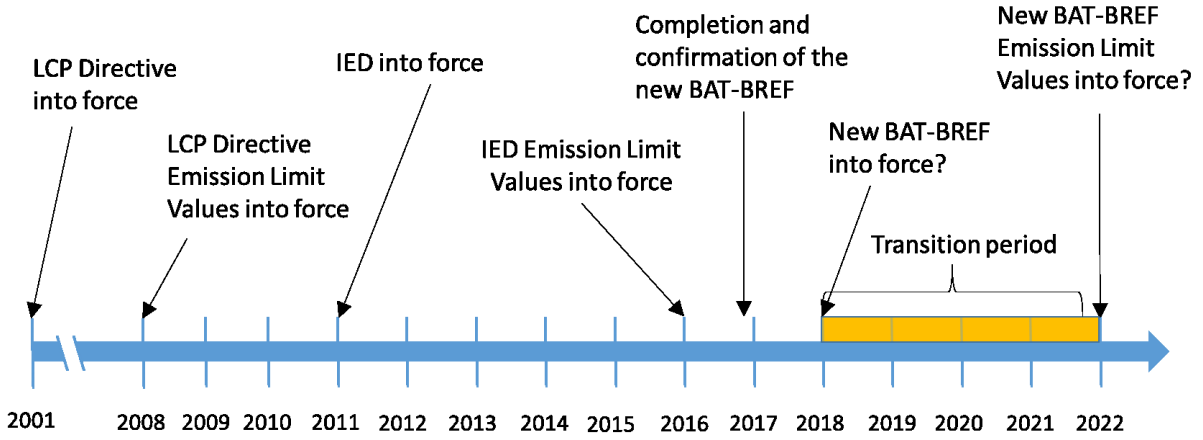

Figure 4: Timeline of the emission regulation process in the EU.

Table 8: BAT associated mercury emissions levels for plants with different fuel and thermal power capacity. (European IPPC Bureau 2016)

Plant fuel and thermal power capacity	Measuring period	BAT-AEL for new plants (µg/Nm^3)	BAT-AEL for existing plants plants (µg/Nm^3)
Coal < 300 MW_{th}	Yearly average or average of samples obtained during one year	< 1-3	< 1-9
Coal ≥ 300 MW_{th}	Yearly average or average of samples obtained during one year	< 1-2	< 1-4
Lignite < 300 MW_{th}	Yearly average or average of samples obtained during one year	< 1-5	1-10
Lignite ≥ 300 MW_{th}	Yearly average or average of samples obtained during one year	< 1-4	< 1-7
Biomass or peat	Average of samples obtained during one year	< 1-5	< 1-5

6 Mercury emission control technologies

Mercury emission control technologies can be divided in three main groups: pre-combustion technologies, co-benefit technologies and mercury specific technologies. As mentioned in the previous chapters, overall mercury removal efficiency depends much on the mercury speciation in the flue gas. While particulate mercury (Hg(p)) and oxidized mercury (Hg^{2+}) can be removed relatively easily, gaseous mercury (Hg^0) is more challenging to be captured. Thus, the main idea of mercury reduction technologies is usually to increase the contents of Hg(p) and Hg^{2+} in the flue gas, which are then captured in different air pollution control devices. In this chapter, the most common mercury reduction technologies and their removal efficiencies are introduced.

6.1 Pre-combustion technologies

Pre-combustion technologies include the methods designed to prevent the formation of mercury emissions prior to combustion, which can be achieved by decreasing the mercury content of the fuel. There are several coal preparation processes for that purpose, such as coal washing and beneficiation, chemical treatment and coal blending.

6.1.1 Coal washing & beneficiation

Coal washing
Coal washing is used for removing mineral impurities from coal, resulting in decreased sulphur, ash, and potentially, decreased mercury content. Conventional cleaning methods involve series of processes based on screening, gravity separation and dewatering. Conventional coal cleaning removes the mineral fraction from the organic fraction of coal, which reduces the mercury content bound in incombustible mineral materials, such as pyrite. However, this will not reduce the mercury bound into the organic material of coal. (UNEP 2010)

Toole-O'Neil, Tewalt et. al. (2001) collected data on mercury removal efficiency of conventional cleaning methods in commercial cleaning plants or commercial-scale cleaning tests. Total of 24 samples were analysed, and in 20 samples the mercury content was decreased. Average removal rate of 37% was achieved, with a variation of 12-78%. However, in one sample the mercury content remained unchanged and in three samples the content actually increased. In another test quoted in UNEP (2010), 26 bituminous coal samples were analysed to explore the mercury removal efficiency of coal cleaning. Averagely, 21% of mercury was removed, as the values ranged from 3% to 64%. Five of the samples showed no mercury removal by the conventional cleaning methods.

The high variation of removal efficiency is probably caused by the differences in chemical composition of the coal samples. While the mercury bound in the minerals can be removed by conventional cleaning methods, the organically bound mercury remains unremoved. When the ash-forming minerals are removed, the organically bound mercury may increase the relative concentration of total mercury in the coal. (Toole-O'Neil, Tewalt et al. 1999)

The effectiveness of coal washing varies significantly, because the effectiveness of the method depends strongly on the source of coal and the type of mercury in coal. This is why coal washing is not so efficient in low-rank coals, such as lignite and subbituminous coal, where mercury is more likely bound in the organic fraction of coal. To determine the feasibility of coal washing technology, it is necessary to implement case-specific laboratory or pilot-scale tests for each coal type. It has to be mentioned that coal washing not only reduces the mercury content of coal, but also other impurities like sulphur and ash particles. This leads eventually to increased fuel efficiency and decreased boiler erosion, and eventually to decreased operation and maintenance costs. The properties of coal washing are represented in the Table 9.

Table 9: Mercury removal properties of coal washing. 1) (Toole-O'Neil, Tewalt et al. 1999) 2) (UNEP 2010)

Removal efficiency of total mercury	~ 37% (12-78%) [1)] ~ 21% (3-64%) [2)] (bituminous coal)
Situation	- Proven technology, used traditionally for sulphur removal
Advantages	- May remove large amounts of mercury relatively easily - Improves also the overall quality of coal by lowering sulphur and ash content - Increases the fuel efficiency and reduces boiler deterioration → reduced O&M costs
Challenges	- Effectiveness depends highly on the coal composition and the mercury type → high variation in mercury removal efficiency

Coal beneficiation (K-fuel technology)

Coal beneficiation can be seen as a next step from the coal washing, and it can achieve better removal of impurities than coal washing alone. Coal beneficiation is always preceded by coal washing, after which additional processes are implemented. One example is the K-Fuel process, which is a multipollutant control technology developed by a company called KFx. K-fuel improves fuel quality by reducing its mercury, ash, moisture, sulphur and some NOx-precursor concentrations. The technology is mainly targeted for low-rank lignite and subbituminous coals, which have high ash and moisture contents. (Tavoulareas, Jozewicz 2005)

The process diagram of K-fuel technology is shown in the Figure 5. K-Fuel constitutes of two steps: physical and thermal separation. In the physical separation, ash (containing mercury and other pollutants) is removed by gravity separation processes, such as crushing and screening. This is followed by thermal separation, in which coal is processed under high temperature (240 °C) and pressure (33 bar), resulting in vaporization of water and mercury. The water and mercury is then entrained from the process and mercury is captured by a carbon-bed adsorption reactor. Water is recycled and mercury is led to disposal. (Tavoulareas, Jozewicz 2005)

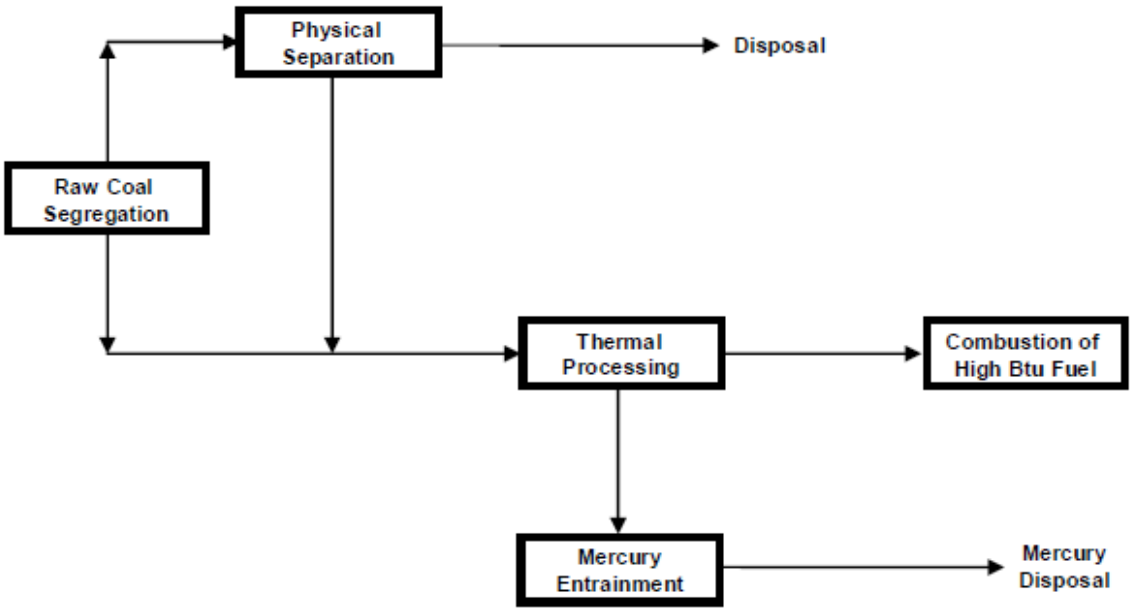

Figure 5: The process diagram of K-fuel technology. (UNEP 2010)

The properties of K-fuel technology are shown in the Table 10. According to KFx, mercury reduction efficiency of 28-66% can be achieved by the physical separation step of the K-fuel process. The thermal separation step has shown a 40% reduction efficiency in some tests. Combining these two steps, a total reduction of 66-67% has been achieved in laboratory tests. It has to be mentioned that K-fuel technology is a multipollutant process which also reduces emissions of SOx, NOx and PM. The reduced moisture content improves the heating value of the fuel and eventually the boiler efficiency. K-fuel process has been tested in small scale facilities and also some commercial plants and the physical and thermal cleaning processes involved in the K-fuel process are well-proven technologies. However, there is still only little operational experience of the process, so he costs of the processed fuel are unclear. (Tavoulareas, Jozewicz 2005).

Table 10: Mercury removal properties of K-fuel technology. *) (Tavoulareas, Jozewicz 2005)

Removal efficiency of total Hg	66-67%*)
Situation	- Emerging, but well-proven technology - Some small scale facilities and commercial plant
Advantages	- Reduces also other emissions, such SO_2 and NOx - Improves the fuel quality by removing ash and moisture → improved efficiency
Issues	- Economic feasibility unclear

6.1.2 Coal blending

Coal blending is a common way to influence the emissions of coal combustion. It has been traditionally used for reduction of SO_2 emissions. By mixing different coal types with each other, it is possible to influence the chemical composition of the fuel and eventually the flue gas composition. In the context of mercury removal, it is favourable to increase the concentration of oxidized mercury in the flue gas, because this can be removed relatively easy in a FGD system. Since halogens, such as chlorine and bromine, promote mercury oxidation, the objective of coal blending is to increase the amount of coal that has a high halogen content (bituminous coal) with coal that has a low halogen content (subbituminous coal). (UNEP 2010)

In some tests made in Holcomb station, USA, bituminous coal was added among subbituminous PRB (Powder River Basin) coal in order to examine the effect of coal blending to mercury removal. The plant had a semi-dry absorber (SDA) and a fabric filter (FF) for emission control. The Figure 6 shows the positive correlation between the share of bituminous coal and mercury removal rate. No removal was occurred when 100% subbituminous coal was used. When the share of bituminous coal was increased to 15%, a mercury removal rate of 80% was achieved. The properties of coal blending are summarized in the Table 11. (Rini, Vosteen 2008)

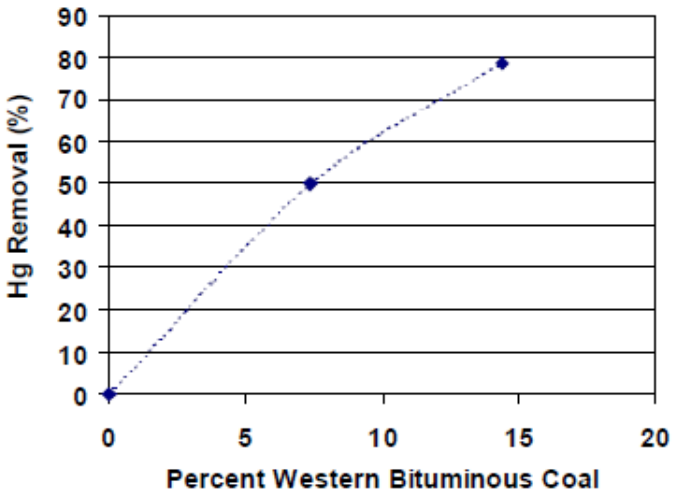

Figure 6: The influence of coal blending on mercury removal at Holcomb Station. (Rini, Vosteen 2008)

Table 11: Mercury removal properties of coal blending. *) (Rini, Vosteen 2008)

Removal efficiency of total Hg	> 80% (with SDA and FF)*)
Situation	- Established technology, traditionally used for SO_2 removal - Mercury removal tests made in commercial power plants
Advantages	- Simplicity, no additional devices needed
Issues	- Site-specific, high-rank coal not always available

6.2 Co-benefit reduction technologies

Co-benefit reduction technologies can be defined as emission control technologies, which are designed primarily for other pollutants than mercury. However, significant mercury reduction rates can also be achieved by these technologies. Co-benefit technologies usually comprehend the most common air quality control devices designed for PM-, NOx- and SOx-emission reduction. Hg^{2+} is water-soluble and can be removed in flue gas desulphurization systems (FGD), whereas Hg(p) can be removed in PM-control devices, such as electrostatic precipitators (ESP) and fabric filters (FF). SCR-systems, which are designed for NOx-reduction, are not able to remove the mercury, but they promote the oxidation of mercury and consequently increase the mercury capture efficiency of a FGD-system.

6.2.1 Particulate matter control technologies

Particulate matter (PM) consist of inorganic impurities which are remained in the flue gas after the fuel combustion. The quantity and size of the particulate matter is determined by the fuel type, combustion technique and the boiler operating conditions. Particulate matter is usually removed by electrostatic precipitators (ESP) and fabric filters (FF). Both of them are very efficient removal technologies, removing over 99 % of the PM. (IEA Clean Coal Centre 2015)

Electrostatic precipitators
The function of electrostatic precipitators (ESP) is based on removing the dust particles by using an electrostatic force. The flue gas is led through an ESP and the particles are negatively or positively charged by discharge electrodes. The particles are then attracted by opposite charged collection plates, on which the particulate matter accumulates. When the thickness of the PM layer is high enough, it is mechanically shaken off by hammers down to a hopper, where it can be gathered and conveyed away. (A. Mizuno 2000)

ESPs can be cold side (installed after air preheater) or hot side ESPs (installed before air preheater). Operation temperatures of cold side and hot side ESPs are 130 – 180 ˚C and 300 – 450 ˚C, respectively. Furthermore, also wet ESPs are sometimes used. In wet ESP, the particles are collected in a liquid film, which is created on the collection plates by spray nozzles. The liquid film makes mechanical rapping unnecessary, and also eliminates the problems with fly-ash resistivity and re-entrainment. The negative side of a wet ESP is the residual waste water and sludge, which requires a proper treatment system. (IEA Clean Coal Centre 2015)

Fabric filters
Another common particulate control device is fabric filter (FF), in which the flue gas is led through long, cylindrical fabric bags, where the particulate matter is filtered. The particulate material is trapped on porous fabric material, forming a so called filter cake. When the dust layer has accumulated enough and the pressure drop is sufficient, a cleaning process will be implemented. There are three main types of cleaning processes, by which fabric filters can be classified: mechanical shaker, reverse-air and pulse jet. In mechanical shaker fabric filters, dust is collected inside the tubular bags, which are cleaned by a shaker mechanism. The dust is shaken off to the bottom of the baghouse and then carried away. In reverse-air fabric filters, dust

is also collected inside the bags, from where it is removed by blowing air temporarily from outside of the bag to the reverse direction. Pulse-Jet fabric filters collect the dust outside of the baghouse, where it is removed by blowing compressed air inside of the bags. (IEA Clean Coal Centre 2015)

Mercury removal

The removal of mercury emissions by PM-control devices is naturally based on particulate mercury (Hg(p)). In order to enhance the mercury capture rate of PM-control devices, it is important to promote the mercury adsorption on the surface of fly ash particles, which increases the amount of Hg(p) in flue gas. This can be done by modification of flue gas or fly ash properties. It is also possible to improve PM-control devices in order to capture particulate matter more efficiently.

Because PM-control devices are able to remove more than 99% of the particulate matter, also about 99% of Hg(p) is removed. The total mercury removal rate depends on the proportion of Hg(p) in the flue gas.

According to UNEP (2010), the reduction rate of ESPs vary between 0 % - 63 %, depending on numerous factors, like the type of fuel burned. Higher reduction rates up to 90% can be achieved by fabric filters. According to Zhang, Wang et al. (2016), on-site measurements have shown an average mercury removal efficiency of 29% in ESPs, with a large range of 1-74%. The efficiency depends also on the boiler type, because the highest rates of 74% have been achieved in circulating fluidized bed boilers. These boilers are more favourable to reactions between the particulate matter and mercury, which leads to higher share of Hg(p).

Flue gas temperature is also an important factor in removal efficiency of ESPs, as lower temperatures seem to enhance total mercury capture. This is because of the fact that gaseous mercury tends to oxidize and condense or adsorb easier onto flue gas particles in cooler temperatures. This in turn leads to higher capture rate in cold ESPs than hot ESPs. Mercury capture rate can be increased by cooling down the flue gas temperature before ESP, but this will also lead to higher risk of SO_3 condensation, which will cause corrosion. (Sloss 2008)

Another way to improve the capture rate is to add halogenated sorbents among the flue gas. Bituminous coals, where the chlorine content is high, are likely to produce more particulate mercury. Also the amount of unburned carbon (UBC) in the flue gas seems to be essential for the formation of Hg(p). Unburned carbon means carbon particles that have not been combusted in the boiler, thus remaining in the flue gas. According to a study of Senior et. al., mercury is more preferentially to be bound in UBC than in the inorganic fraction of the fly ash. UBC is able to both adsorb and oxidize the mercury. Some pilot-scale tests have shown that increase of UBC in the flue gas increases also the content of Hg(p), which naturally results in higher capture rate of ESPs and FFs. However, conclusions cannot yet be generalized to large scale power plants. The increasing trend of mercury capture in ESP as a function of UBC content in the flue gas, as can be seen in the Figure 7. (Senior, Johnson 2005)

ESPs may also have influence in the amount of oxidized mercury. It has been noticed that the charging anode of ESP can convert Hg^{2+} back to Hg^0. On the other hand, mercury oxidation

processes continue inside ESPs. Thus, the Hg^0 content of flue gas can be either higher or lower after an ESP. (Zhang, Wang et al. 2016)

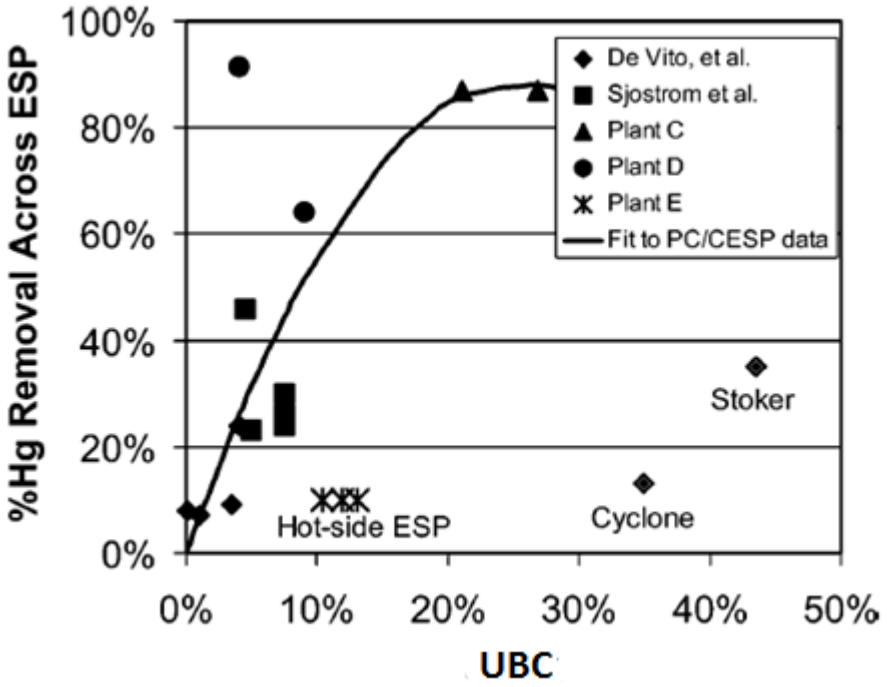

Figure 7: Mercury capture in ESP as a function of unburned carbon (UBC) content in flue gas. (Senior, Johnson 2005)

Fabric filter has turned out to be more effective in terms of mercury control, because the filter cake acts as a fixed-bed reactor, offering more contact surface between particulate matter and mercury. This facilitates both mercury adsorption and oxidation, increasing the amount of Hg^{2+}. The average mercury removal efficiency of a FF is estimated to be 67 % with a range of 9 – 92%. The filter cake can remove over 50% of Hg^{2+}. (Zhang, Wang et al. 2016)

Field test were made by Wang, Duan et al. (2008) in coal-fired power plants in China in order to explore the removal efficiency of ESPs and fabric filters. Tests were made in five commercial power plants with capacities ranging between 50-600 MW. Plants 1 and 2 were equipped with FF and plants 3, 4 and 5 with ESP. The plants 1 and 2 showed removal rates of 80% and 20%, and the plants 3, 4 and 5 removal rates of 6%, 20% and 4%, respectively. Especially the removal rate in plants 1 and 2 differed significantly from each other, although both of them were equipped with FF. Also, the removal rates of plants 3 and 5 differed clearly from plant 4. The reason for this was supposed to be variation on flue ash composition, such as unburned carbon content and alkaline metal oxides.

Summary

Mercury removal properties of electrostatic precipitators and fabric filters are summarized in Tables 12 and 13, respectively. It can be concluded that fabric filters are more efficient than ESPs in terms of mercury removal. One explanation for this is higher interaction between particulate matter and mercury in fabric filters. Furthermore, especially hot side ESPs operate

usually in higher temperatures, which is not favourable for the mercury capture from the fly ash. (Srivastava, Hutson et al. 2006)

Table 12: Mercury removal properties of ESP. 1) (Zhang, Wang et al. 2016) 2) (Wang, DUAN et al. 2008)

Mercury removal efficiency	~ 29% (1-74%) [1)] 4-20 % [2)]
Situation	Established and wide-spread technology
Advantages	Low investment costs
Challenges	Conversion of Hg^{2+} to Hg^0 by the charging anode
Sources:	

Table 13: Mercury removal properties of FF. 1) (Zhang, Wang et al. 2016) 2) (Wang, DUAN et al. 2008)

Mercury removal efficiency	~ 67% (9-92%) [1)] 20 – 80% [2)]
Situation	Established technology
Advantages	Filter dust cake promotes mercury oxidation
Challenges	More expensive than ESP

6.2.2 Selective Catalytic Reduction (SCR)

Selective Catalytic Reduction (SCR) is a common technology to control nitrogen oxides (NOx) -emissions in power plants. It has been commercially in use since 1980s and the installations are growing due to introduction of more stringent NOx-limits in many countries. NOx-emission reduction rates of 80 – 90% can be achieved by SCR technology. SCR has also been discovered to oxidize mercury effectively. Most studies have shown that the oxidation rate can be as high as 85% - 90%. It shall be noted that SCR does not remove mercury itself, but the oxidized mercury can be removed in downstream FGD-system. (IEA Clean Coal Centre 2015, Srivastava, Hutson et al. 2006)

SCR system consists of reactor, catalysts and ammonia storage and injection system. The catalysts inside the reactor are the key element for NOx reduction. As the name suggests, the reduction of nitrogen oxides (NOx) is based on chemical reactions on the catalysts in a presence of reducing agent, usually ammonia vapour. The process in a SCR reactor is illustrated in the Figure 8. In the process, ammonia (NH_3) vapour is injected into the flue gas before the catalyst by an ammonia injection system. The flue gas is then directed through the catalyst blocks, where ammonia reduces NOx-compounds into molecular nitrogen (N_2) and water (H_2O). Nitrogen and water exit the reactor among the flue gas. For a proper function, the temperature of flue gas has to be between 300 - 400°C. The main reactions on the catalyst are (Miller 2004):

$$4NO + 4NH_3 + O_2 \rightarrow 4N_2 + 6H_2O \quad (3)$$

$$2NO_2 + 4NH_3 + O_2 \rightarrow 3N_2 + 6H_2O \quad (4)$$

The catalysts are usually made of vanadium and titanium, vanadium being the active metal and titanium the supporting mesh. Also zeolite, iron oxide or activated carbon can be used, and the catalyst consists usually of many active metals to meet specific requirements of SCR. For the catalyst layers, there are mainly two different geometries in use: flat plate and honeycomb. A plate type catalyst produced by Mitsubishi Hitachi Power systems is shown in the Figure 9. Some experiences have shown that flat plates are more resistant to clogging and erosion. (IEA Clean Coal Centre 2015)

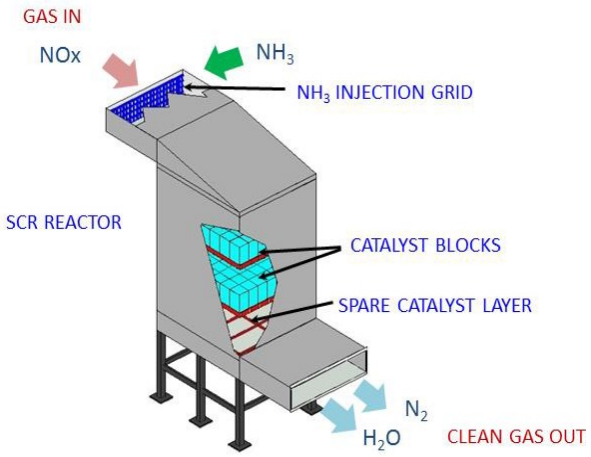

Figure 8: NOx reduction process in a SCR reactor. (Ekman 2016)

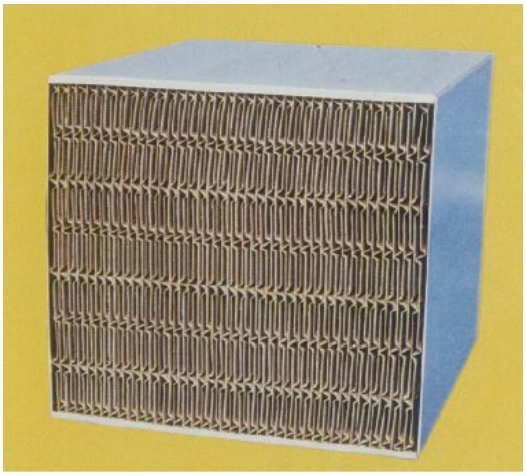

Figure 9: Plate type catalyst used in an SCR. (Favale, Nakamoto et al. 2013)

Mercury oxidation

Mercury oxidation over a SCR catalyst has been tested in many laboratory and pilot plants, but still some of the process mechanisms remain unknown. Although over 85-90% oxidation rates have been achieved in many studies, it has been noticed that mercury oxidation rate varies widely and depends on numerous factors. According to Zhang, Wang et al. (2016), some field tests have shown an average oxidation rate of 71% with a range of 34-85%. The fuel type, and consequently the chemical composition of the flue gas, have a significant effect on the oxidation process. According to He, Zhou et al. (2008), the oxidation range is about 30-98% for bituminous coal and 0-26% for subbituminous coal.

It has been proven that chlorine is an essential chemical for an efficient mercury oxidation in the SCR catalysts. Chlorine, usually in the form of HCl (g), is naturally present in coal or injected together with a sorbent to the flue gas stream. Chlorine is absorbed in the vanadium oxide (V_2O_5) catalyst, where it reacts with gaseous Hg^0 in the catalyst and forms $HgCl_2$. Another explanation proposes that reaction occurs via so called Deacon Reaction, in which gaseous chlorine (Cl_2) is formed from the HCl through a catalytic reaction on the V_2O_5 -layer. The gaseous chlorine (Cl_2) then oxidizes the Hg^0 in the flue gases. The Deacon reaction is presented as: (He, Zhou et al. 2008)

$$4HCl\ (g) + O_2\ (g) \rightarrow 2Cl_2\ (g) + 2H_2O\ (g) \quad (5)$$

A study conducted by He, Zhou et al. (2008) examined the mercury adsorption over a SCR catalyst in the presence of HCl. Tests were made by directing flue gas through four different configurations and measuring the share of oxidized mercury from the total mercury concentration after the flue gas had passed the configuration. The four configurations were: 1. Injection of HCl without an SCR, 2. SCR without injection of HCl, 3. Injection of HCl with SCR and 4. Injection of HCl with SCR, but Hg^0 was brought separately to the flue gas after the SCR reactor. The results are shown in the Figure 10.

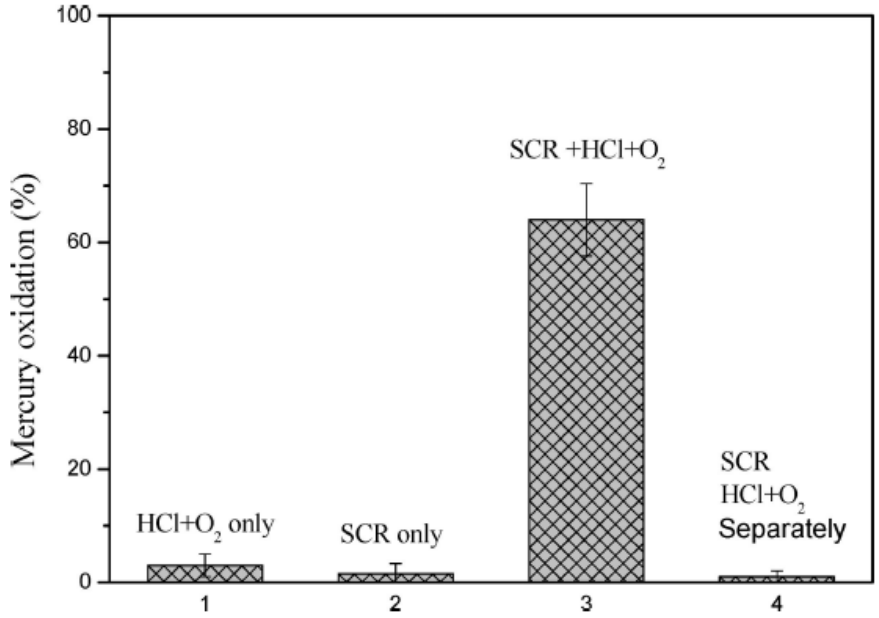

Figure 10: Mercury oxidation rates in the presence of SCR and HCl. (He, Zhou et al. 2008)

The results show clearly that presence of HCl in the SCR process is very effective in regard to mercury oxidizing. As configurations 1, 2 and 4 led to very low oxidation rates, in configuration 3 a total oxidizing rate of 64% was achieved. Conclusions were that without the SCR catalyst, the reaction barrier between HCl and Hg^0 is too high for proper mercury oxidation. In the configuration 4, it seems that the gaseous chlorine do not contribute to the mercury oxidation after the SCR reactor. The main conclusion from this study was that HCl adsorbs on the catalysts, which leads to the oxidation of mercury in the catalyst. (He, Zhou et al. 2008)

Another study made by Li, He et al. (2012) researched also the mercury oxidation efficiency over a fresh and used catalyst at different temperature rates and HCl concentrations. The used catalyst had been already 40 000h in operation. The study measured oxidation rate in two different conditions: oxidation condition (N_2+O_2+HCl+Hg+SCR), where no nitrogen oxides nor ammonia was present, and SCR condition (N_2+O_2+HCl+NO+NH_3+Hg+SCR), where nitrogen and ammonia were involved. The results can be seen in Figures 11 and 12. (LI, HE et al. 2012)

The importance of HCl was confirmed also in this research. The oxidation rate has positive correlation with the HCl concentration. A total of 100% oxidation rate was achieved for the fresh catalyst, and 82% for the 40 000h catalyst in ideal conditions. In oxidation condition, both catalysts seemed to reach a saturation level at relatively small amount of injected HCl (4μg/m^3),

after which no significant increase in mercury oxidation was reached anymore. However this was not the case in SCR condition, where NH_3 and NO were present. The oxidation rates are significantly lower in SCR condition than in oxidation condition. This implies that NH_3 might compete with HCl in the catalyst adsorption, which results in decreased HCl adsorption and Hg oxidation. (LI, HE et al. 2012). Nevertheless, some field tests have shown that NH_3 may actually promote mercury oxidation, which leaves the role of NH_3 controversial (Zhang, Wang et al. 2016).

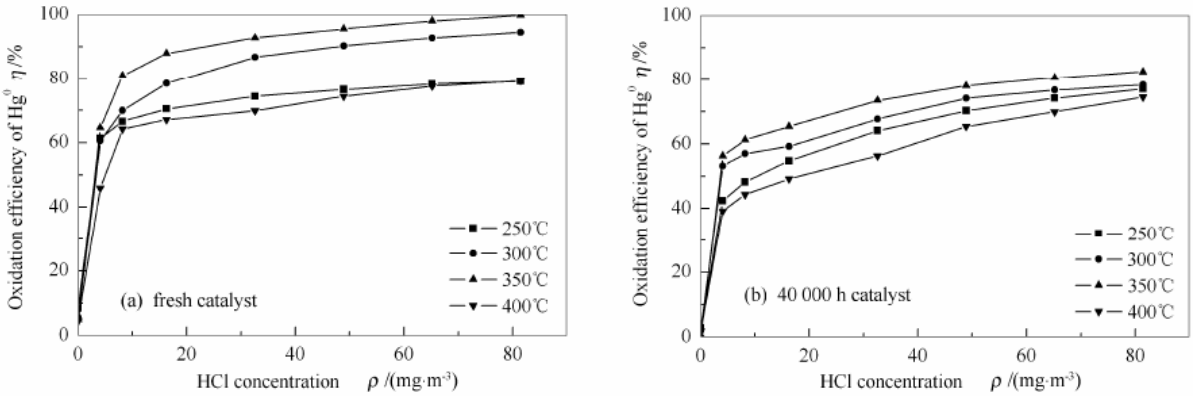

Figure 11: Mercury oxidation efficiency in a) fresh catalyst and b) 40 000h SCR catalyst as a function of HCl-concentration and flue gas temperature. Oxidation mode (NH_3 and NO not present). (LI, HE et al. 2012)

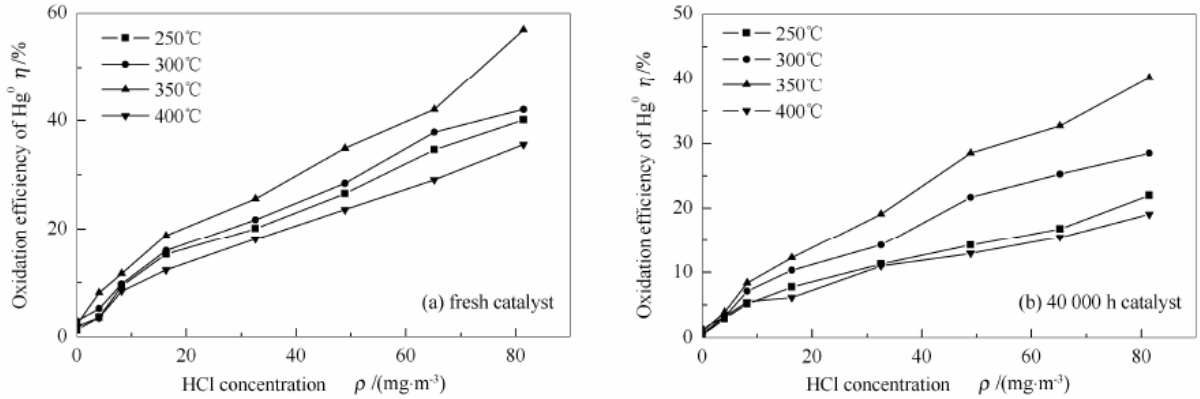

Figure 12: Mercury oxidation efficiency in a) fresh catalyst and b) 40 000h SCR catalyst as a function of HCl-concentration and flue gas temperature. SCR-mode (NH_3 and NO present). (LI, HE et al. 2012)

The influence of temperature is also shown in the Figures 11 and 12. In all conditions, an optimal temperature for both fresh and 40000h catalyst was found to be 350 ˚C, where the oxidation efficiency peaked clearly. The oxidation efficiency was about 20% lower at temperatures 250 ˚C and 400 ˚C. (LI, HE et al. 2012)

The oxidation efficiency of the catalyst deteriorates slowly during its lifetime, as can be seen in the Figures 11 and 12. The decrease is about 5-20% between the fresh and 40000h catalyst. There are several reasons for this. Firstly, fly ash deposits on the surface and holes of the catalyst

after a long operation time, which reduces the diffusion of Hg^0 and HCl, and the chemical reactions between them. Secondly, the vanadium content of the catalyst seemed to be lower in the used catalyst. Also, the higher contents of alkali metal ions (Na, K, Ca and Mg) in the 40000h catalyst decreased the oxidation performance. (LI, HE et al. 2012)

The effect of sulphur oxides on the oxidation efficiency of a catalyst has been investigated in several studies. It has been noticed that SO_2 can either promote or inhibit the oxidation, depending on the flue gas conditions. In the absence of oxygen, Hg^0 oxidation is inhibited by SO_2, but in the presence of oxygen, small concentrations of SO_2 seem to promote the oxidation. In high O_2 concentrations, the effect is again inhibiting. The reason for this is supposedly SO_3, which has a beneficial effect on Hg^0 oxidation. Presence of O_2 contributes to oxidation of SO_2 to SO_3, which also increases the Hg^0 oxidation. There seems to be a balance point in SO_2 concentration, where the inhibiting effect of SO_2 and promoting effect of SO_3 are equal. Additionally, presence of NO catalyses the formation of SO_3, which raises the SO_2 balance point. (Li, Wu et al. 2013)

In a case where HCl is present, SO_2 is considered as inhibitor, because it competes with HCl in oxidation. Also, HCl is much more effective oxidizer than SO_3, so the promoting effect of SO_3 is negligible. In this case, the presence of NO is also desired, because it promotes SO_2 oxidation to SO_3, thus diminishing effect of SO_2 to Hg^0 oxidation. (Li, Wu et al. 2013)

TRAC®-catalyst

TRAC® (Triple Action Catalyst) is a special catalyst for mercury removal developed by Mitsubishi Hitachi Power Systems. It is especially designed for high Hg^0 oxidation, but it also lowers SO_2 oxidation and maintains the NOx reduction high, thus the name Triple Action. TRAC® seems also to be more resistant for NH_3, which is used for NOx reduction. (Favale, Nakamoto et al. 2013)

After several research, pilot and field test, TRAC has been commercially available since 2007. Tests conducted in USA and Europe have shown that TRAC increases the mercury oxidation. A full-scale test was performed in a power plant combusting bituminous coal, where one of the three conventional catalyst layers was replace with TRAC. Installation of TRAC increased the ratio of Hg^{2+}/Hg^0 from 40% to 70% at a downstream FGD inlet. This resulted in an increase in the total mercury removal from 30% to 70%. (Favale, Guglielmo et al. 2011)

In particular, TRAC is suitable for combustion of low rank coals, which have usually poor halogen content. With a conventional SCR catalyst, the absence of oxidizing halogens leads to low formation of Hg^{2+}. By a TRAC, this can be improved without adding extra oxidizing chemicals. Instead, combustion of high-chlorine coal forms normally enough Hg^{2+} also with conventional SCR catalysts. (Favale, Guglielmo et al. 2011)

Summary
To summarize, the main factors having an effect on the mercury oxidation are

- HCl, NO, SO_2 and NH_3 concentration in the flue gas
- temperature of the flue gas
- catalyst age.

It has been proven that HCl oxidizes effectively mercury to $HgCl_2$, thus being a critical component for the mercury oxidation. The role of excess NH_3, which is used for NOx control, remains controversial. Some research articles suggest that NH_3 inhibits mercury oxidation, while other articles indicate that it may actually promote the oxidation. The effect of SO_2 is also controversial. Depending on the other flue gas components, it can either promote or inhibit the oxidation. Too high or two low temperature may decrease the oxidation efficiency. Also, the catalyst age has significant influence, because catalyst oxidation performance deteriorates slowly in the operation. The mercury oxidation properties of SCR are shown in the Table 14.

Table 14: Properties of mercury oxidation in SCR. 1) (Zhang, Wang et al. 2016) 2) (He, Zhou et al. 2008)

Mercury oxidation efficiency	71% on average, (34-85%)[1)] 30-96% for bituminous fuel[2)] 0-26% for subbituminous fuel[2)]
Situation	- Established technology
Advantages	- Promotes mercury oxidation
Challenges	- Possible adverse effect of NH_3 and SO_2 on mercury oxidation

6.2.3 Flue gas desulphurization systems

Flue gas desulphurization (FGD) systems comprehend a variety of technologies developed for sulphur dioxide (SO_2) removal. The first FGD systems were installed in some coal-fired power plants and industrial installations in the early 1970s in the United States and Japan, followed by a rapid expansion in Europe in the 1980s. FGD systems can be classified in several ways. A common way is a division into three categories: Wet flue gas desulphurization (WFGD), semi-dry absorption (SDA) and dry sorbent injection (DSI), which can be further divided into different subcategories. Two main FGD systems, that are relevant in mercury removal, are introduced in this chapter: Wet flue gas desulphurization (WFGD) and semi-dry absorption (SDA). (Córdoba 2015, Zwers 2011)

Wet flue gas desulphurization (WFGD)
Wet flue gas desulphurization systems are the most popular technology for SO_2 removal, mainly because of the high desulphurization performance and low operating costs. As high as 92 - 98% SO_2 removal rate can be achieved by WFGD. The process uses alkaline slurry, such as limestone ($CaCO_3$) or slaked lime ($Ca(OH)_2$), as a sorbent for SO_2 removal. Another sorbents are ammonia and sodium based alkali solutions. Limestone is usually preferred due to its availability and relatively low cost. (Córdoba 2015)

A limestone-based WFGD process is described in the Figure 13. In the process, flue gas is directed into an absorber where the reactions take place. Limestone slurry is injected in the absorber through spray nozzles. The spray nozzles disperse the limestone slurry into tiny droplets, maximizing the reaction surface between the slurry and the flue gas. As a result, sulphur dioxides are absorbed into the limestone slurry and a series of complicated reactions are occurred. The simplified main reaction between sulphur dioxide (SO_2) and limestone ($CaCO_3$) can be expressed as (Miller 2004):

$$SO_2\,(g) + CaCO_3\,(s) \rightarrow CaSO_3\,(s) + CO_2\,(g) \quad (6)$$

The main products are calcium sulphite ($CaSO_3$) and carbon dioxide (CO_2). Some of the calcium sulphite is naturally oxidized into calcium sulphate ($CaSO_4$), which can be used as gypsum. The end products have low water solubility and are naturally precipitated on the bottom of the absorber. Because gypsum has economic value, the rest of the $CaSO_3$ is usually also oxidized to $CaSO_4$ through a mechanical oxidation. The simplified chemical reaction of the oxidation of calcium sulphite can be expressed as: (Córdoba 2015)

$$2CaSO_3\,(aq) + 4H_2O\,(l) + O_2\,(g) \rightarrow 2CaSO_4 \cdot 4H_2O\,(s) \quad (7)$$

In mechanical oxidation, the final end-product can contain 90% of $CaSO_4$. The gypsum is then removed from the process for dehydration, thickening and storage. The unreacted limestone slurry can be recirculated back to the absorber by a recirculation system and mixed with fresh limestone slurry. Additional process water is led into the absorber in order to replace the water that is removed along with the gypsum. (Córdoba 2015)

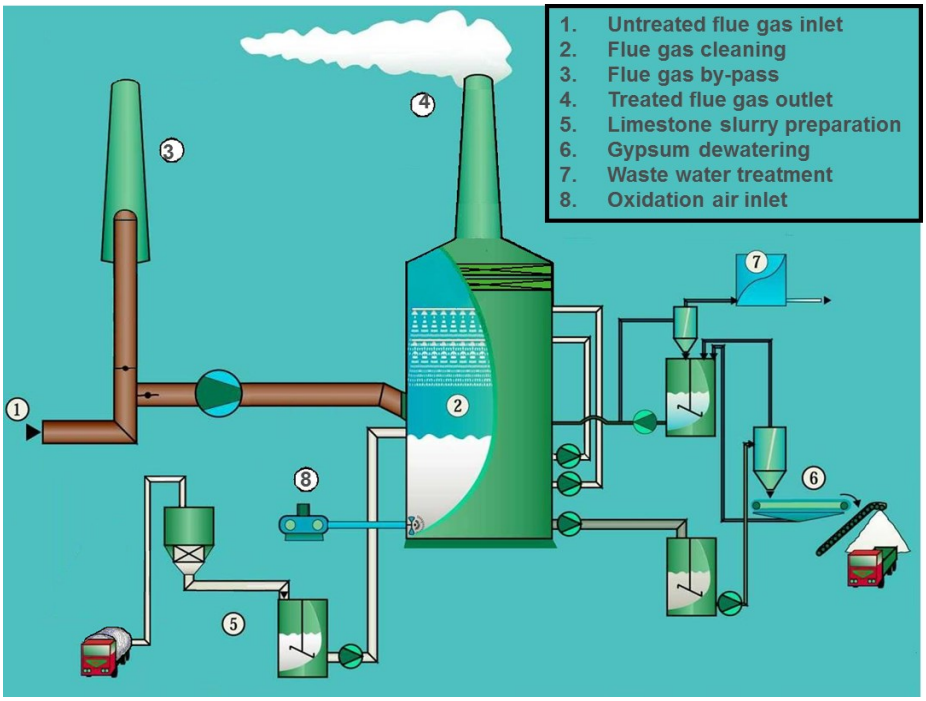

Figure 13: Limestone based wet FGD system. (Ekman 2015)

Semi-dry absorption (SDA)

Semi-dry absorbers are the second most used technology for SO_2 removal. Removal rates from 85% to 95% can be achieved by this technology. Semi-dry absorption can be seen to comprehend the following three technologies: Spray dry absorber (SDA), circulating fluidized bed (CFB) scrubber and Novel Integrated Desulphurization (NID). (Zwers 2011)

The basic idea of all three semi-dry absorption technologies is similar. Semi-dry absorbers use slaked lime ($Ca(OH)_2$) as a sorbent to capture SO_2-emissions. The flue gas is humidified with water to cool down the gas in order to enhance the desulphurization. The difference to wet FGD is that the water is completely evaporated by the flue gas heat, and no waste water is produced. Dry calcium sulphate and sulphite are formed as main products. The main reactions are (Córdoba 2015):

$$Ca(OH)_2 + SO_2 \rightarrow CaSO_3 + H_2O \quad (8)$$

$$Ca(OH)_2 + SO_3 \rightarrow CaSO_4 + H_2O \quad (9)$$

As an example of a semi-dry absorption process, spray-dry absorption is described in the Figure 14. In a spray-dry absorption process, slaked lime slurry is prepared at the site by mixing lime with water. The slurry is injected in the absorber, and the by-products, as well as unreacted lime and fly ash, are collected in a downstream particulate control device, such as ESP or FF. (Zwers 2011)

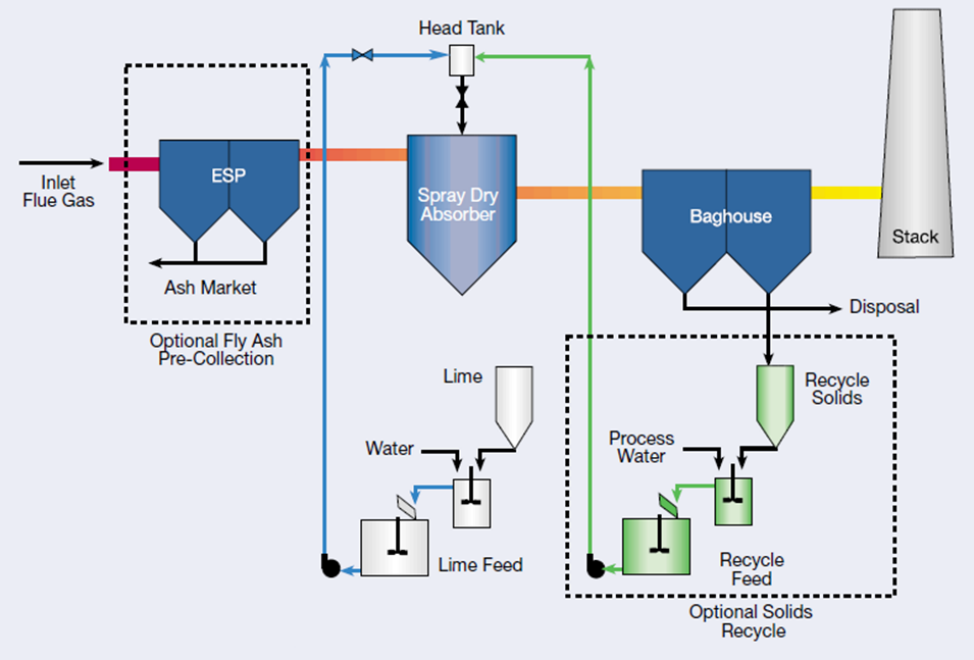

Figure 14: Spray-dry absorption process. (Ekman 2015)

The unreacted lime can be recycled and mixed with the fresh lime in order to reduce lime consumption. Rest of the by-products are disposed. Critical parameters in spray-dry absorption are correctly sized lime slurry droplets and proper residence time in PM-control devices, so that the particulates are dry but well reacted. (Córdoba 2015)

The CFB-process is similar to spray dry absorber process, but part of the fly ash, partially reacted sorbent and by-products are recirculated from the PM-control device back to the absorber in large quantities. This allows the sorbent to react several times with the flue gas. A part of the by-products are discharged from the absorber and the PM-control device to a by-product silo. Fresh lime and water are injected in the recirculation process to replace the spent sorbent. Different to spray dry absorption, hydrated lime is introduced to the process in a granular form with very little excess water, which enables a better adjustment of lime injection according to SO_2 concentration in the flue gas. (Buecker, Hovey 2011)

In the NID-process, which is developed by Alstom, the reactions take place in a narrow J-tube reactor. The different to other semi-dry technologies is that the NID is an entrainment process, so there is no absorber or fluidized reactor. The reactions have to occur fast, as the flue gas spends only about one second in the reactor. This is ensured by hydrated lime ($Ca(OH)_2$), which is produced from quick lime and water in a humidifier. The very small particle size of hydrated lime enables the quick reactions. One advantage of NID-process is a modular design, which makes it easier to make retrofit installations in existing power plants. (Buecker, Hovey 2011)

The use of semi-dry absorbers is usually limited to small and medium sized coal fire plants, about 200 MW as an average capacity, while wet FGD systems are more suitable to larger power plants. One advantage compared to wet FGD is that no waste water treatment system is needed in semi-dry absorber systems, which saves investment costs. Also, the absorber can be made of cheaper materials due to less corrosive operating conditions. However, semi dry absorption consumes more sorbent, which increases the operating costs. (IEA Clean Coal Centre 2015)

It has to be mentioned that in various literature articles that originate from US, the term spray dry absorber seems to be used instead of semi-dry absorption. It remains unclear, whether it refers to only spray dry absorbers or semi-dry absorbers in general. In this study, the term semi-dry absorption (SDA) is thought to cover all the semi-dry absorption technologies, assuming that the results of mercury removal tests made by spray dry absorbers apply to semi-dry absorption technologies in general.

Mercury removal

Dry and wet FGD-systems have a significant role in mercury removal. The oxidized form of mercury (Hg^{2+}) is water-soluble and can be captured effectively in an existing FGD system among the scrubber solution. Because chlorine is the dominant oxidizer in coal derived flue gases, Hg^{2+} usually appears as a mercury chloride ($HgCl_2$). FGD systems are not able to capture Hg^0, and it passes through the scrubber.

According to Zhang, Wang et al. (2016), the total mercury removal rate of a wet FGD system is about 64 % with a range of 56-88 %. The values are based on data collected from eight different sources. The removal efficiency is predetermined by the mercury speciation, composition of the

incoming flue gas and FGD system operating parameters, such as pH, temperature and chemical composition of the slurry. (Díaz-Somoano, Unterberger et al. 2007)

Díaz-Somoano, Unterberger et al. (2007) have studied the influence of FGD scrubber parameters on mercury removal efficiency. In their experiments, a 75% mercury removal was observed in average. As one of the main results, a significant correlation between mercury removal efficiency with SO_2 concentration and pH was found. The mercury removal efficiency increases with the pH value and decreases with the SO_2 concentration. The reasons for this seems to be that mercury tends to react with sulphate-ions (SO_4^-) to form mercury sulphate ($HgSO_4$). $HgSO_4$ is one of the main products formed in the Hg-scrubber solution. A high pH value ensures the oxidation of sulphite, which increases the interaction between Hg and sulphate-ions. Furthermore, higher slurry concentration increases mercury removal. This is probably due to increased amount of calcium (Ca) particles, which favours the adsorption of Hg.

One drawback of mercury removal in FGD systems is the re-emission phenomenon. Some amount of the already captured Hg^{2+} tends to be reduced back to Hg^0 and re-emitted into the atmosphere from the flue gas stack. Despite of researches, the details of the re-emission reactions are not fundamentally known. Some parameters, such as temperature, pH and sulphide, oxygen and chloride concentrations seem to have influence in the re-emission. High temperature and pH are connected with higher vapour pressure of elemental mercury in the scrubber solution, which increases the re-emission. It is believed that re-emission is caused by formation of complexes between Hg^{2+}, sulphite, chloride and possibly thiosulphate ions. Anyway, re-emissions seem to be a cause of several complex chemical processes in the absorber. (European IPPC Bureau 2016). (Reissner, Crèvecoeur et al. 2015, Lipinski, Leonard et al. 2011)

The mercury removal efficiency of an FGD relies on the minimizing of re-emissions. There are several oxidizing additives that have been designed to prevent mercury re-emission. The basic idea is to precipitate oxidized mercury from the scrubber solution before it reacts with reducing chemicals, such as sulphite ions. In some cases sodium hydrosulphide (NaHS) seems to hinder the conversion when injected into the scrubber solution. Halogens, such as chlorine and bromines, prevent also the re-emission by forming complex salts with mercury. In this state, mercury is in a more stable form. Another way to reduce re-emissions is addition of activated carbon, which adsorbs mercury efficiently. (Ghorishi, Downs et al. 2006, Reissner, Crèvecoeur et al. 2015)

Several field tests have been made in order to investigate the mercury removal efficiency of wet FGD systems in varying conditions. In the review of Srivastava, Hutson et al. 2006, five tests in three power plants have been introduced. All of the tests had different configurations. Test configurations and the results of the test can be found in the Table 15.

Table 15: Results of the field tests of the review made by Srivastava, Hutson et al. (2006).

Power plant	Fuel	Capacity	Reduction technology	Oxidized (Hg^{2+}) mercury removal	Total mercury removal
Endicott Station (Michigan, USA)	High-sulphur bituminous coal	55 MW	WFGD + NaHS	96%	76 – 79%
Zimmer Station (Ohio, USA)	High-sulphur bituminous coal	1300 MW	WFGD + NaHS	87%	51%
Dominion Resources power plant (West Virginia, USA)	Medium-sulphur bituminous coal	563 MW	WFGD	>90%	71%
Dominion Resources power plant (West Virginia, USA)	Medium-sulphur bituminous coal	563 MW	WFGD + NaHS	>90%	78%
Dominion Resources power plant (West Virginia, USA)	Medium-sulphur bituminous coal	563 MW	WFGD + SCR	>95%	>90%

Almost all of the tests resulted in 70% total mercury removal. Zimmer Station seems to be an exception. Although 87% of Hg^{2+} was removed, only 51% of total mercury removal was achieved. This indicates that the NaHS-injection didn't work successfully in this case, which resulted in an increase of 40% of Hg^0 content across the WFGD. Most interesting case was the Dominion Resources power plant, where the effect of NaHS and SCR can be clearly seen. Without NaHS and SCR, the total mercury removal was 71% and Hg^{2+} removal >90%. When NaHS was added, mercury capture increased to 78% and Hg^{2+} removal was about the same, >90%. This indicates that NaHS effectively prevented the reemission. When the flue gas was directed through the SCR, total mercury reduction increased significantly, even though no NaHS was added. This indicates that the use of SCR may also prevent mercury reemission in some cases. (Srivastava, Hutson et al. 2006)

Combinations of SDA and FF can remove about 95 % of the mercury, when burning bituminous coal. The oxidized mercury is finally removed by FF. However, the capture rate seems to be smaller, when subbituminous coal is used. Tests have shown that in some cases mercury capture can be smaller with SDA/FF configuration than in a plant with only FF. The reason for this seems to be the SDA, which also scrubs the chlorine and other halogens that are important for the mercury oxidation. As mentioned in the chapter 5.2.1, fabric filter can promote the oxidation of mercury, because the filter cakes offer reaction surface for mercury and oxidizers. (Srivastava, Hutson et al. 2006). Mercury removal properties of FGD systems are represented in the Table 16.

Table 16: Mercury removal properties of FGD systems. 1) (Zhang, Wang et al. 2016). 2) (Díaz-Somoano, Unterberger et al. 2007). 3) (Srivastava, Hutson et al. 2006)

	WFGD	WFGD + NaHS	WFGD + SCR	SDA +FF
Mercury removal efficiency of total mercury	45% (10-85%) [1)] 75% [2)] 71% [3)]	51-79%[3)]	>90% [3)]	95% [3)]
Mercury removal efficiency of Hg^{2+}	>90% [3)]	87-96% [3)]	>95% [3)]	
Situation	- Well established technology			
Advantages	- Cost-effectiveness - Simultaneous reduce of sulphur oxide emissions - Production of marketable gypsum as a by-product (WFGD only)			
Challenges	- Mercury re-emission issues - Investment costs are high for WFGD			

6.3 Mercury specific reduction technologies

Mercury specific technologies can be used when co-benefit reduction technologies alone do not offer high enough mercury removal rate. The methods introduced in this category are based on sorbent injection and/or halogen addition into the flue gas or fuel, which increases the share of Hg(p) and Hg^{2+}. Mercury specific reduction technologies are used in conjunction with FGD, SCR and PM-control devices.

6.3.1 Sorbent injection

One of the most common mercury-specific reduction technologies is sorbent injection into the flue gases. The purpose of the sorbent is to promote adsorption and oxidation of the elemental mercury (Hg^0), which increases the shares of Hg^{2+} and Hg(p) in the flue gas. These can be then removed in downstream PM-control devices (ESP or fabric filter) and FGD systems. The most used sorbent is powdered activated carbon (PAC) and the process is called activated carbon injection (ACI). ACI has been proved to be an effective way to reduce mercury emissions. Depending on the coal type, up to 90% removal rates can be achieved by ACI. The performance of ACI can be increased by chemical treatment. (Granite, Pennline et al. 2014)

Mercury capture by activated carbon is based on mercury adsorption on activated carbon surface. The large surface are of the activated carbon particles is the key factor for efficient mercury oxidation and adsorption processes. Powdered activated carbon is produced by processing carbon with high temperature steam, which results in a porous structure with high surface area. Mercury is adsorbed on the PAC and captured along with the fly ash in a downstream ESP or FF. (Shewchuk, Azargohar et al. 2016)

Sorbent injection is a simple process that can be retrofitted easily in existing plants. The process is described in the Figure 15. In a typical configuration, sorbent is injected into the flue gas duct

between the air preheater and the particulate control device (ESP or FF). Activated carbon injection system consists of a storage silo, conveying belt and an injection system. Also a metering system is required to monitor the consumption of the sorbent. PAC is transferred from the storage silo usually by screw feeders into a drop tube, from which it is fed to the injection system. In the injection system, the PAC is pneumatically discharged into the flue gas duct through manifolds and individual injection lances. (Moretti, Jones et al. 2012)

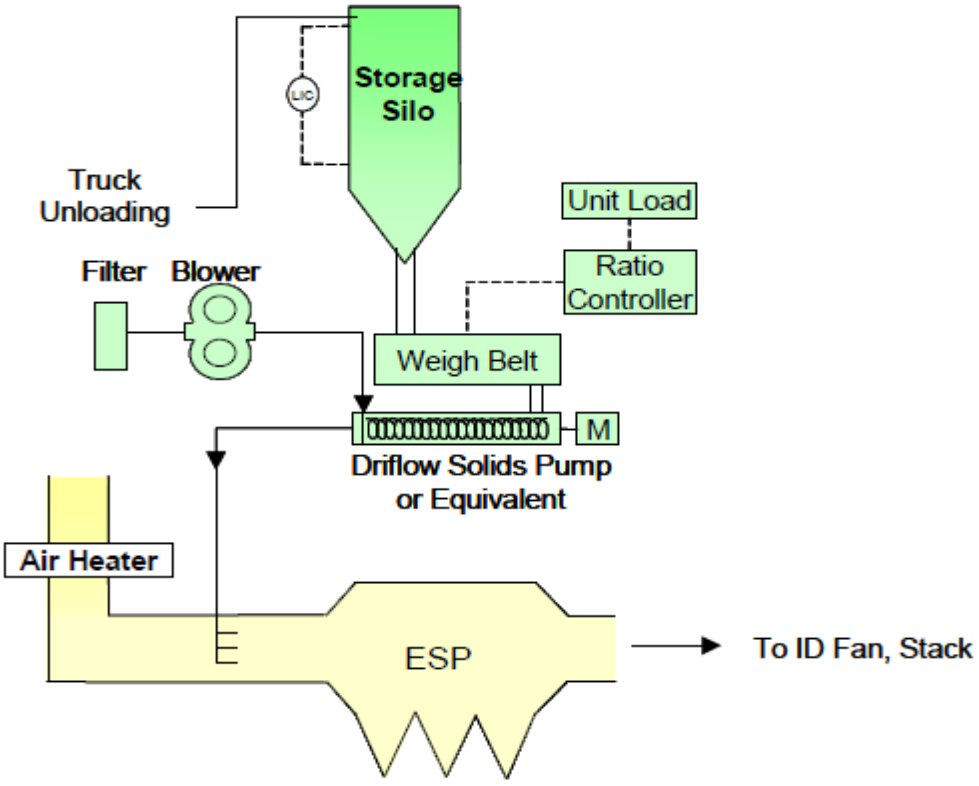

Figure 15: Schematic of activated carbon injection system. ESP = Electrostatic precipitator. (Lipinski, Leonard et al. 2011)

Mercury adsorption by activated carbon is a complex process. The removal efficiency depends at least on the following factors (Srivastava, Hutson et al. 2006):

- Method and rate of activated carbon injection.
- Physiochemical characteristics of the activated carbon, such as porosity and particle size.
- Flue gas conditions and temperature.
- Concentrations of halogen species (e.g. Cl and Br) and sulphur trioxide (SO_3).
- Existing air pollution control configuration.

Mercury is assumed to be adsorbed onto activated carbon by two ways: physisorption and chemisorption. Physisorption is physical mercury adsorption on the micropores of the activated carbon. Physical adsorption is much more dependent on temperature than chemisorption. High

contact probability, high surface area of activated carbon and sufficient residence time are prerequisites for an efficient physisorption. (Reissner, Crèvecoeur et al. 2015)

In chemisorption, mercury forms chemical bonds with so called functional chemical agents on the surface of activated carbon particles. These kind of functional chemical agents are HCl, O_2 and SO_2 molecules. Especially sulphur has been noticed to be essential for efficient mercury adsorption, and in some conditions, sulphur coated carbons have been noticed to be 40 % more efficient than non-sulphur coated carbons. Although sulphur may decrease the micropore surface area, it seems offers to offer excellent sites for mercury chemisorption. (Shewchuk, Azargohar et al. 2016)

Nevertheless, the role of sulphur is controversial. While the presence of SO_2 tends to enhance mercury capture, SO_3 might suppress it. This is a problem especially with medium or high sulphur coals. As a result of the inhibiting effect, PAC consumption increases. According to some tests, in SO_3 concentrations of 15 ppm PAC becomes much less effective. In order to mitigate this problem, it is possible to use lime or soda for SO_3 removal before the PAC injection. (Moretti, Jones et al. 2012, Shewchuk, Azargohar et al. 2016)

Halogens, such as chlorine, bromine and iodine are also efficient functional agents that promote mercury oxidation and chemisorption. Especially chlorine and bromine are commonly used to enhance the mercury capture of activated carbon. As a result, lower injection rates are required to achieve the same mercury removal rate. Because bromine is more effective than chlorine, it is the most used halogen. It is not certainly known why bromine is more efficient, but that is supposed to arise from different oxidizing mechanisms. Bromine seems also be more stable than chlorine. The downside of the bromine is a higher cost. (Moretti, Jones et al. 2012, Shewchuk, Azargohar et al. 2016).

Temperature is also one determining factor in mercury adsorption process. Tests indicate that the efficiency decreases as the temperature increases. According to Moretti, Jones et. al (2012), efficiency of untreated activated carbon starts to decrease as temperature goes above 175 °C, but the limit can be raised up to approximately 200 °C by halogenation. Physisorption is assumed to occur primarily at low temperatures (~ 50 °C) and chemisorption is favoured in higher temperatures (>150 °C) in the presence of functional agents. Most studies have pointed out that the optimal temperature range for chemisorption processes is from 120 to 180 °C. (Shewchuk, Azargohar et al. 2016)

As the content of halogens is a significant factor in mercury oxidation, the removal efficiency of activated carbon depends much on the fuel type. In plants combusting high-chlorine bituminous coal, the removal efficiency can be 90% by normal activated carbon, when a SCR is a part of the pollution control system. When subbituminous coal is combusted in a plant, the removal efficiency may be only 60%. However, by halogenated activated carbon the removal rate can be increased up to 90 % also with low-rank coals. While injection of untreated activated carbon seems to work well only in plants burning bituminous coal, chemically treated activated carbon is effective also in plants burning subbituminous coal or lignite. (Moretti, Jones et al. 2012)

A comprehensive, three-phase field testing program on mercury removal by activated carbon injection has been made by National Energy Technology Laboratory (NETL), which belongs to U.S. Department of Energy (DOE). The first phase began in 2000 by exploring the removal rates of untreated activated carbon injection, followed by the second phase started in 2003, which explored the removal rates with chemically enhanced activated carbon. (Feeley III, Jones et al. 2009)

The coal types that were used in the tests were lignite and Powder River Basin (PRB) subbituminous coal, so the both coal types can be regarded as low rank coals. Electrostatic precipitators (ESP) and fabric filters (FF) were used to remove particulate matter. The amount of injected activated sorbent has been expressed as kilograms per million actual cubic meters of flue gas (kg/MMm^3), which equals milligrams per actual cubic meter (mg/m^3).

The results of these tests are shown in the Figure 16, which describes the removal efficiency of activated carbon injection with different coal types and particulate matter control devices. In general, remarkable improvements in mercury capture are achieved by ACI. During the first phase, only untreated activated carbon injection was explored. In a lignite fired plant using a fabric filter, a removal efficiency of 60 % was achieved by an activated carbon injection rate of 50 kg/MMm^3. Also, the other plants reached 50 - 60 % removal efficiency by higher injection rates. However, no significant improvement were gained by additional injection rates. This is obviously due to low chlorine content of subbituminous coal, which is essential for the mercury oxidation and adsorption on the activated carbon particles. Additionally, high calcium and sodium content of subbituminous coal may also neutralize the halogen species in the flue gas. (Feeley III, Jones et al. 2009)

The second phase of the tests explored the effect of chemically treated activated carbon. Significant improvements can be made by treated ACI, as can be seen in the Figure 16. A reduction efficiency of 90 % at an injection rate of 50 kg/MMm^3 was achieved in all plants regardless of the coal type. It can also be seen that higher removal efficiencies were achieved when ACI was used in combination with fabric filter. However, if the injection rate was low or zero, electrostatic precipitator seems to be more effective. (Feeley III, Jones et al. 2009)

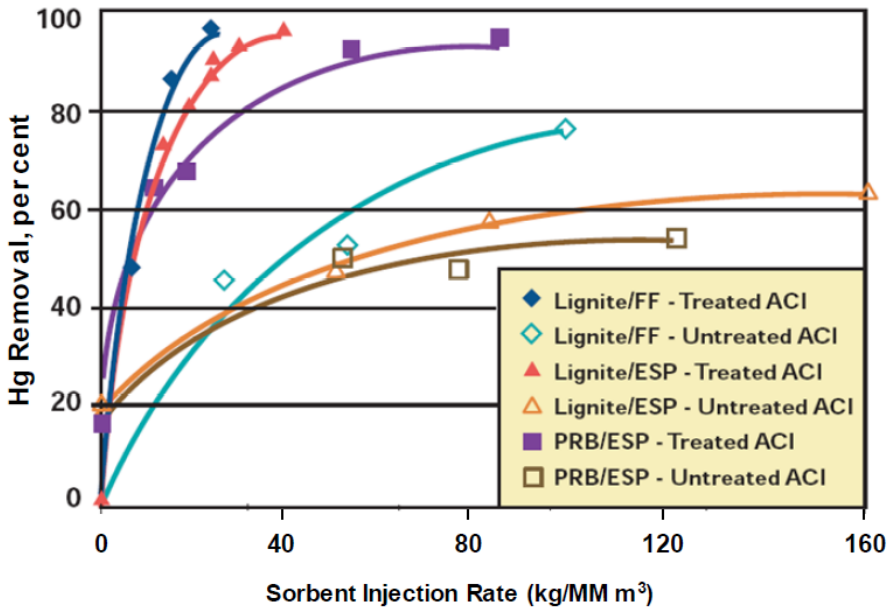

Figure 16: Mercury removal efficiency of untreated and treated activated carbon injection (ACI) with different coal types and particulate matter control devices. FF = Fabric filter, ESP = Electrostatic precipitator, PRB = Powder River Basin (subbituminous coal type in the U.S.). (Feeley III, Jones et al. 2009)

The properties of sorbent injection are summarized in the Table 17. Sorbent injection has been recognized to be a cost-effective way to provide additional mercury removal. One concern regards the possible corrosion in the flue gas duct and other metal surfaces caused by some halogen-treated activated carbons. Especially bromine has been noticed to accelerate corrosion in some tests. Researches are under way to explore the long-term corrosive impacts of halogenated sorbent injection. Furthermore, because the sorbent injection system is located upstream to particulate matter control devices, the sorbent and the fly ash are removed as a mixed composition. Because of the mixing, there has been some concerns regarding the fly ash quality. Fly ash can be used as commercial product cement production, but it sensitive to the carbon content of the ash. However, sorbent producers have already been developing ash-friendly sorbents that should not affect the fly ash quality. Furthermore, fly ash quality problems can also be avoided by separating the fly ash before sorbent injection. This idea is applied in TOXECON process, which is introduced in the chapter 6.3.3. (Granite, Pennline et al. 2014)

Table 17: Mercury removal properties of sorbent injection. 1) (Moretti, Jones et al. 2012)

Mercury removal efficiency	- 60% without chemical additives (halogen additives)[1)] - 90% with chemical treatment (halogen additives)[1)]
Situation	- Commercially promising technology for mercury removal - Currently a standard practice
Advantages	- Low investment costs - Simple installation
Challenges	- May affect the fly ash quality - Adverse effect of SO_3 on the mercury adsorption - Limited temperature range (especially with untreated sorbent) - Corrosion issues with brominated PAC

6.3.2 Halogen additives

The oxidation of Hg^0 can also be promoted by injection of oxidizing chemicals into the fuel. Common chemicals are halogen salts based on chlorine or bromine, such as calcium chloride ($CaCl_2$) or calcium bromide ($CaBr_2$). Especially bromine has been proven to be a cost-effective way to oxidize mercury. These salts vaporize during the combustion and form HBr, Br_2, HCl and Cl_2, which can oxidize mercury into Hg^{2+}. Chlorine and bromine can also be found in natural coal, but their concentrations may not be high enough to ensure sufficient oxidation. This is especially the case in low rank coals, such as lignite and subbituminous coal. (Granite, Pennline et al. 2014)

The equipment consists of a bulk storage tank, metering pumps and an injection system. Basically, the chemicals are added as an aqueous solution. The solution is pumped from the storage tank into a pipeline and injected through a simple lance, which is an open-ended steel tube without a nozzle. An even distribution of the chemical is not necessary, since the coal will be mixed during the pulverization process. (Granite, Pennline et al. 2014)

There are many options for the injection point of halogens. The typical halogen injection points are shown in the Figure 17. The easiest application is to inject chemicals into the fuel prior to combustion. The most common injection point is at the coal feeders, just before pulverizers. Another option is to inject the chemicals directly into the boiler during the combustion. Furthermore, it is also possible to inject halogens into the flue gas duct. Flue gas injection installation requires a grid injection system, and a proper mixing of the chemical has to be assured. However, after-combustion installations may be more costly. In general, more downstream injection point will lead to higher costs, because lower temperatures will usually lead to decreased efficiency. This means higher consumption rate and more costs. On the other hand, more downstream injection point leads to faster injection control response, because the emission measurement system is located in the flue gas stack. (Granite, Pennline et al. 2014)

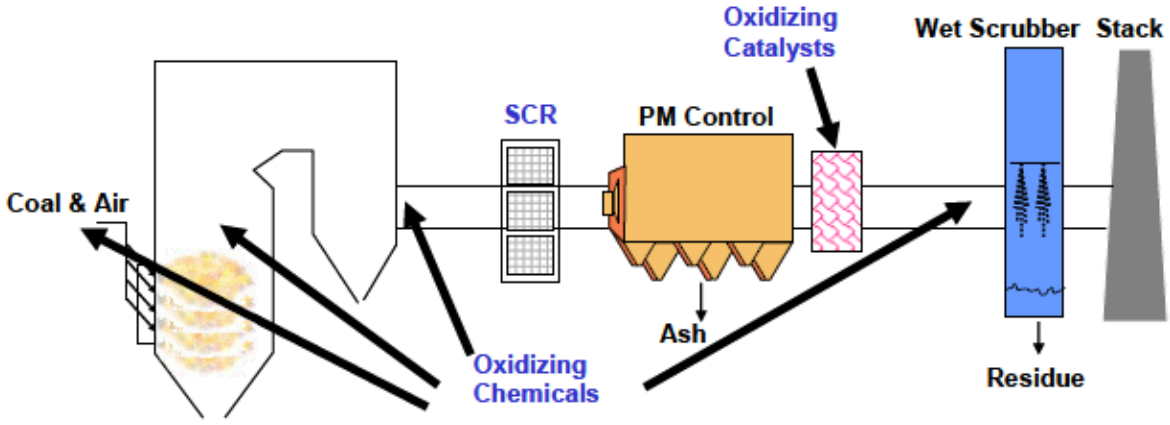

Figure 17: Options for injection points of halogen additives. (Strivastava 2010)

The oxidizing effect of halogens and consequent improvement in mercury removal can be seen in the Figure 18. It is notable that bromine injection of 100 µg per one gram of coal (100 ppm) results in 80 % mercury removal, while a 10-fold amount (1000 ppm) of chlorine injection results only in 60% removal rate. The removal efficiency of halogen addition depends also much on the existing flue gas cleaning configuration. High oxidation efficiencies are achieved with significantly lower injection rates, if a SCR is present. In general, if SCR is applied, high enough oxidation is usually achieved with bromine injection rate of 20-25 ppm. As a comparison, if SCR is not applied, 100-150 ppm injection rate is needed. (Granite, Pennline et al. 2014, Lehmkuhler 2016).

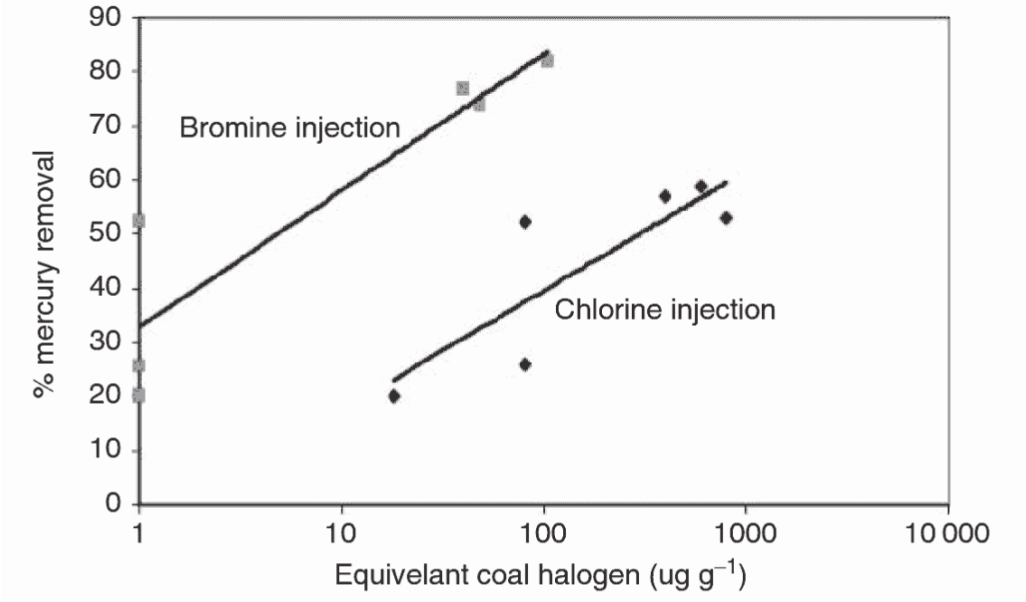

Figure 18: The effect of halogen addition on mercury removal in coal fired boilers. (Granite, Pennline et al. 2014)

Some full-scale tests have been conducted in a number of power plants burning subbituminous and lignite coal. In these tests, addition of 20-200 ppm bromine resulted in 50-90% oxidation rates. As an example, bromine injection tests were conducted at Monticello plant, USA. The plant is burning 50/50 blend of subbituminous and lignite coal and it is equipped with an ESP and wet scrubber. Mercury removal rates were increased from 10-40 % to 65 % with bromine addition of 55 ppm. Furthermore, increasing the injection rate to 113 ppm, resulted in removal efficiency of 86%. (Leonard et al. 2011)

Alstom has developed a bromine addition technology known as KNX technology. In KNX process, bromine containing chemical is added into the coal prior to combustion. Test conducted by Alstom, U.S. Electric Power Research Institute (EPRI) and Department of Energy (DOE) have shown a mercury removal efficiencies more than 90%. Tests were made by 600 MW coal-fired unit with selective catalytic reduction (SCR), electrostatic precipitator (ESP) and wet flue gas desulphurization (WFGD). The technology has also been in commercial utilization in some German waste and sludge incineration plants. (Rini, Vosteen 2008). The properties of halogen are shown in the Table 18.

Table 18: Mercury removal properties KNX technology. 1) (Rini, Vosteen 2008). 2) (Granite, Pennline et al. 2014). 3) (Leonard et al. 2011).

Removal efficiency of total Hg	> 90% (KNX with SCR, ESP and WFGD)[1)] 80% (Bromine injection)[2)] 86 % (Bromine injection with ESP and WFGD)[3)]
Situation	- Emerging technology - Full-scale tests made in commercial power plants
Advantages	- Low investment costs, - Easy installation - Efficiency
Challenges	- Relatively high costs of bromine - Bromine may be corrosive

6.3.3 TOXECON

TOXECON (Toxic Emission Control Process) is a patented mercury removal method developed by EPRI (The Electric Power Research Institute). The process was developed to overcome the issues with the decrease of fly ash marketability when using sorbent injection. In TOXECON, the sorbent injection system is installed between ESP and secondary PM control-device, usually a fabric filter. The configuration of TOXECON process can be seen in the Figure 19. First, the fly ash is collected by the ESP-devices, after which the sorbent is injected into the flue gas. The spent sorbent, which reacts with the mercury, is then removed in a pulse jet fabric filter. (Feeley III, Brickett et al. 2005)

The configuration makes it possible to avoid the mixing of fly ash and sorbent, because the fly ash is collected in the ESP before the sorbent injection, so the fly ash quality can be maintained.

Also, the overloading of ESP-devices by the injected sorbent can be avoided. Additionally, pulse jet fabric filter has been proved to require less sorbent than ESPs in order to achieve the same mercury removal efficiency, which is a cost-saving factor. On the other hand, fabric filter increases investment costs. (Muggli, Durham et al. 2006)

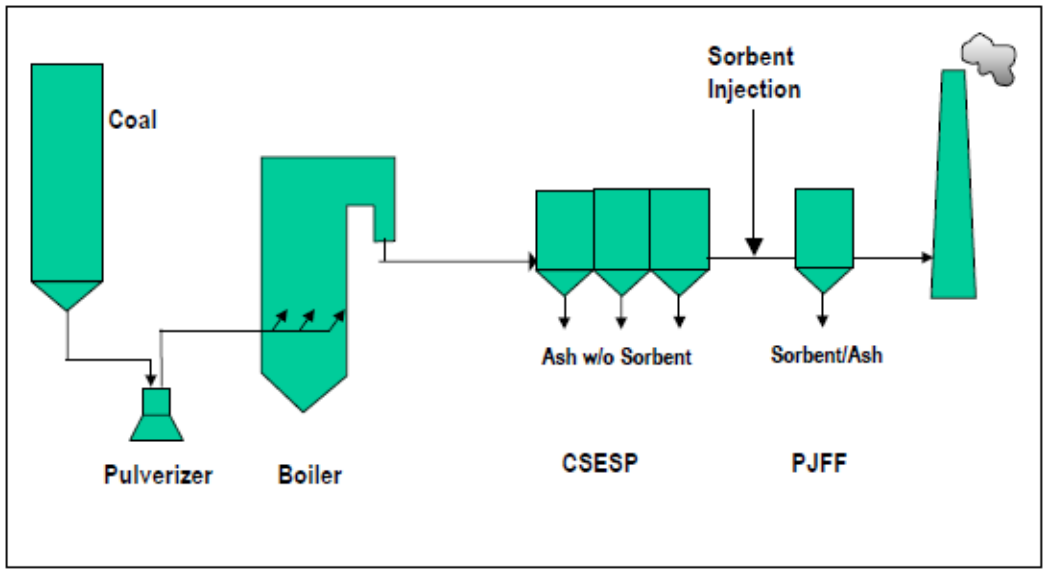

Figure 19: Configuration of TOXECON process. (Feeley III, Brickett et al. 2005)

In order to reduce the additional investment cost caused by the fabric filter, another version of TOXECON process has been developed. In TOXECON II, the sorbent injection is installed in the middle of ESP. The majority of the fly ash is collected in the first layers of ESP and only a small fraction is mixed with sorbent, which is finally removed before the flue gas exits in the last layers of ESP. By this configuration, investment in a fabric filter system can be avoided. (Feeley III, Brickett et al. 2005)

The removal efficiencies of both TOXECON processes have been under examination in full-scale power plants. TOXECON I has been utilized in Presque Isle Power Plant (Michigan, USA), which has been commercially operational since 2006. A removal efficiency of 90 % total mercury was achieved during 48 consecutive days with both untreated and brominated sorbent. The injection rate of the sorbent was less than 50mg/Nm^3. TOXECON II has been tested in Independence Station Unit 1 (Arkansas, USA) and again, 90 % removal efficiency has been achieved. In this case, the sorbent injection rate has been bigger, about 90 mg/Nm^3. (Feeley III, Jones et al. 2009). The mercury removal properties of TOXECON process are shown in the Table 19.

Table 19: mercury removal properties of TOXECON process. 1) (Feeley III, Jones et al. 2009)

Removal efficiency of total Hg	90%[1)]
Situation	- TOXECON I: commercial phase - TOXECON II: field tests
Advantages	- Does not affect the fly ash quality
Challenges	- TOXECON I requires installation of another PM-control device

6.4 Summary of mercury emission control technologies

Some of the most common mercury removal technologies and their removal efficiencies were introduced in this chapter. Several researches and field tests have shown that mercury can be removed effectively by existing air pollution control devices, such as FGD, SCR, ESPs and FFs. In principal, there are two ways to remove mercury from the flue gas: Either removing particulate-bound mercury (Hg(p)) by using particulate control devices or capturing oxidized mercury (Hg^{2+}) in FGD systems.

SCR does not remove mercury, but it may facilitate the mercury oxidation. For example, with a combination of SCR and WFGD, over 90% mercury removal rates can be achieved with bituminous coals. Formation of Hg(p) and Hg^{2+} can be further promoted by mercury specific technologies, such as sorbent and halogen injection. This enhances the overall mercury removal of the plant. Another solution to increase the share of Hg^{2+} are coal washing, beneficiation and blending.

The behaviour of mercury in a typical flue gas cleaning configuration is described in the Figure 20. In general, the mercury control strategy should be based on maximizing the concentrations of Hg(p) and Hg^{2+}, because they can be easily removed in flue gas cleaning devices. Other way around, Hg^0 is challenging to be captured by any flue gas cleaning device, which makes it an unfavourable form of mercury. The optimal solution for mercury removal is to maximize the utilization of co-benefit technologies, such as FGD, SCR and PM-control. This leads also to reduction of other pollutants. If these technologies do not provide sufficient mercury removal rates, additional mercury specific control equipment can be applied.

The efficiency of different mercury control technologies depends a lot on the mercury speciation in the flue gas. These in turn depend on site-specific conditions, combustion environment, type of fuel burned and other boiler configuration. For example, a power plant burning high-chlorine fuel tends to achieve a higher mercury removal rate because of higher amount of oxidized mercury. Accurate efficiency estimates for the technologies cannot be given, and the estimation has to be made case-specifically for each plant. The effect of coal type can be seen in the Figure 21. The data is collected by US EPA from different power plants (Strivastava 2010). In the plants that burn bituminous coal, more than 90% of the mercury can be captured in flue gas

cleaning configurations. On the other hand, in some plants that burn lower rank coals, hardly any of the mercury has been able to be removed.

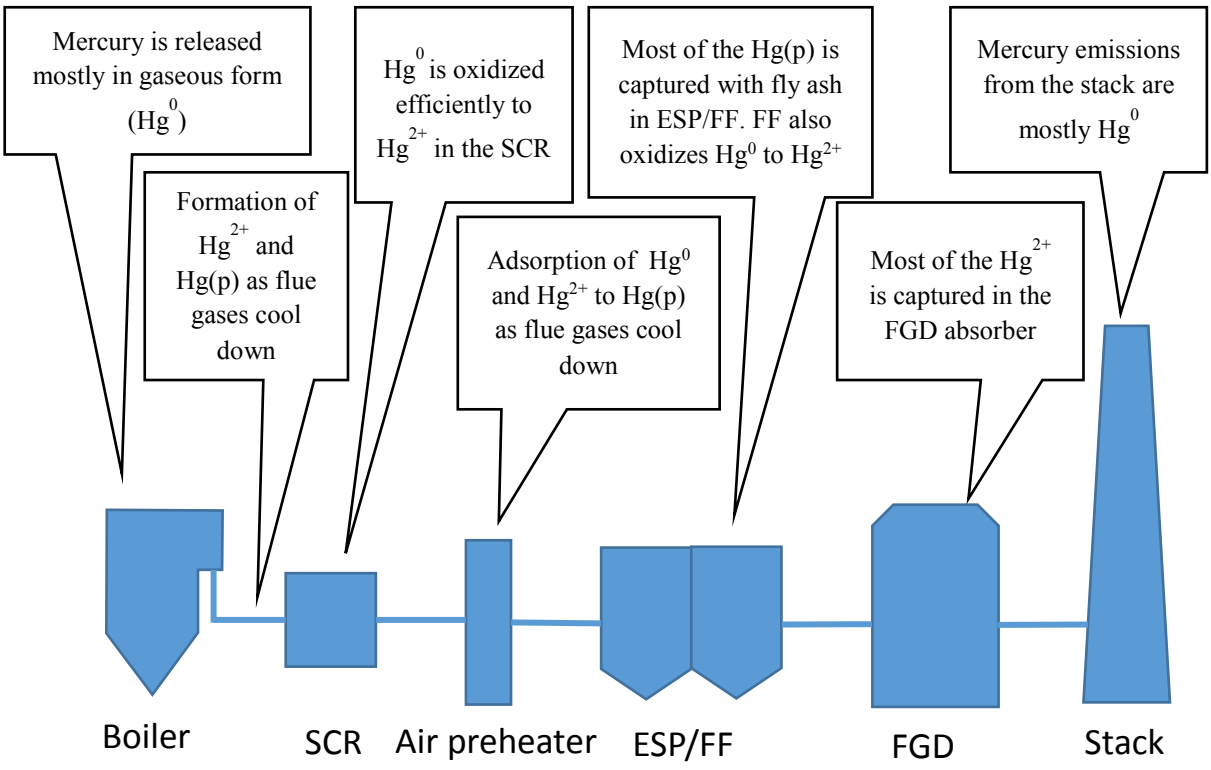

Figure 20: Behaviour of mercury in a typical flue gas cleaning configuration.

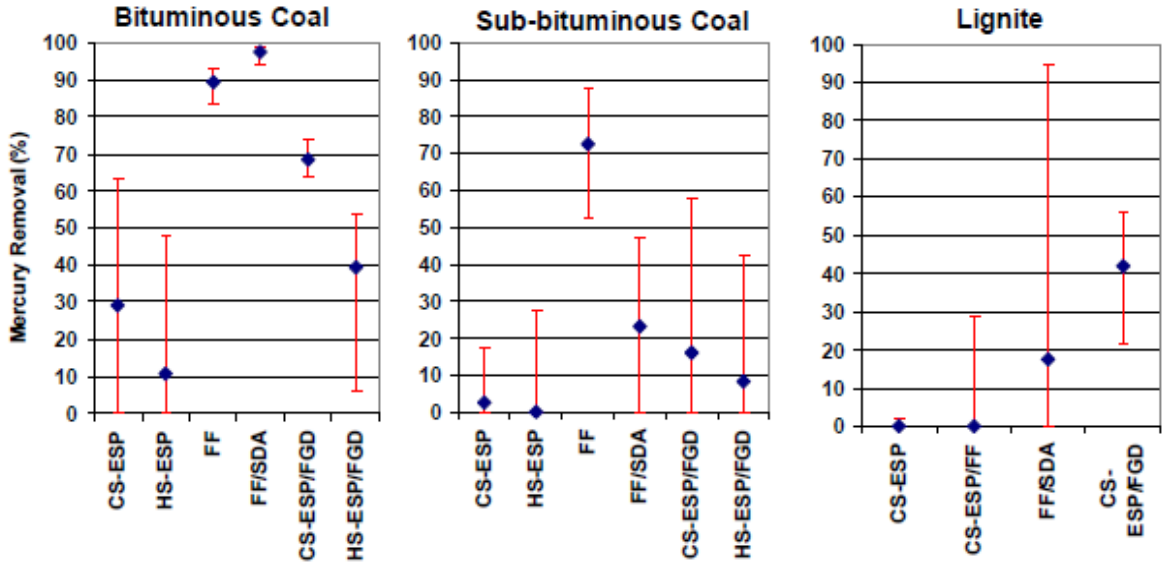

Figure 21: Mercury removal efficiency for three coal types in different flue gas cleaning configurations. (Strivastava 2010)

Many literature sources offer data for mercury removal efficiencies in different flue gas cleaning configurations. The removal efficiencies reported in these sources are either based on field tests or collected from other literature sources. Table 20 summarizes some data about mercury removal efficiencies collected from different sources. Results are reported for each coal type, or for all coal types, if the coal type is not specified. As can be seen in the Table 20, mercury removal efficiencies for high rank bituminous coal are significantly higher than those for lower rank subbituminous or lignite coal. Another common observation is that the mercury removal in fabric filter is more efficient than in electrostatic precipitator for all coal types.

Table 20: Estimated mercury removal efficiencies with different coal types by different flue gas cleaning configurations.

Configuration	Bituminous coal (%)	Subbitumi-nous coal (%)	Lignite (%)	All coals (%)	Remarks
PM control					
ESP					Good capture of Hg(p)
CS-ESP	22[8)] 29[5)] 36[1)] 36[4)] 56[2)]	3[4)] 3[5)] 9[1)] 12[2)]	0[5)] 1[1)] 47[2)]	28[3)] 29[9)] 29[6)] 42[2)]	Good capture of Hg(p)
HS-ESP	9[4)] 11[5] 14[1)] 27[2)]	0[5] 6[4)] 7[1)] 9[2)]		20[2)] 28[9)]	Lower capture of Hg(p) due to high temperature
FF	85[2)] 89[5)] 90[8)] 90[1)]	72[1)] 73[5)] 75[2)]	58[2)]	67[6)] 67[3)] 67[9)] 82[2)]	Good co-benefit capture, may oxidize Hg^0 to Hg^{2+}
PS		9[1)] 9[4)]		22[9)] 23[6)] 26[2)]	
SO_2 controls					
WFGD				64[6)]	
PM and SO_2 controls					
SDA+ESP		35[4)] 43[1)] 53[2)]		53[2)]	
SDA+FF	83[2)] 98[1)] 98[4)]	22[2)] 24[4)] 25[1)]	2[1)] 25[2)]	53[2)] 59[9)]	Very good co-benefit capture for bituminous coals, less effective for lower rank coals
PS+WFGD	12[1)] 12[2)] 12[4)]	10[1)] 18[2)]	33[4)]	16[2)]	

CS-ESP+WFGD	51[2)] 69[5)] **70[8)]** 74[4)] 81[1)]	16[5)] 27[2)] 29[4)] 29[1)]	42[5)] 44[4)] 48[1)] 48[2)]	45[2)] 62[9)] 64[3)]	Good capture of mercury for bituminous coals because of the Hg^{2+}, poorer capture for lower rank coals
HS-ESP+WFGD	39[5)] 46[1)] 50[4)]	8[5] 20[1)] 35[2)]	29[4)]	35[2)] 40[9)]	Moderate capture for bituminous coals, poor capture for lower rank coals
FF+WFGD	75[5)] 98[1)] 98[4)] 98[8)]		73[2)]	73[2)] 86[3)] 90[9)]	Good co-benefit capture, FF may increase the amount of Hg^{2+}
NOx, PM and SO_2 controls					
SCR + SDA + FF	97[5)] 98[1)] 98[4)]	23[5)]	17[5)]		Very good co-benefit capture expected for bituminous coal, less for lower rank coals. SCR enhances the capture by oxidizing Hg^0 to Hg^{2+}.
SCR+CS-ESP+WFGD	85[8)]			66[9)] 69[3)]	
SCR+FF+WFGD				90[3)]	High capture expected for all coal types.
Nox, PM, SO_2 and Hg controls					
SCR+FF+ACI +WFGD				97[3)]	Very high mercury capture due to sorbent injection
SCR+ESP+ACI-FF+WFGD				99[3)]	Very high mercury capture due to sorbent injection
HI+SCR+ESP +WFGD				95[3)]	Very high mercury capture due to halogen injection

ESP = Electrostatic precipitator, CS-ESP = Cold side ESP, HS-ESP = Hot side ESP, FF = Fabric filter, PS = Particle scrubber, SDA = Semi-dry absorber, SCR = Selective catalytic reduction, WFGD = Wet flue gas desulphurization, ACI = Activated carbon injection, HI = Halogen injection.

Sources: 1) (Yudovich, Ketris 2005), 2) (Hepola 2003), 3) (Ancora, Zhang et al. 2015), 4) (Srivastava, Staudt et al. 2005), 5) (Srivastava, Hutson et al. 2006), 6) (Zhang, Wang et al. 2016) 7) (Pavlish, Hamre et al. 2010) 8) (Wang, Zhang et al. 2010) 9) (UNEP 2011)

RESEARCH PART

7 Fundamentals of cost analysis

7.1 Basic theory of interest

Interest can be considered as a time value for money. The value of a certain amount of money is greater now than the value of the same amount of money in the future, because the money can be invested in order to make profit. In other words, the amount of money invested today leads to increased amount of money in the future as a result of interest. In investment calculations, interest is used for expressing the return requirement for an investment. The general rule of interest rate can be expressed as (Luenberg 2009):

$$V = (1 + rt)A \tag{10}$$

Where,

A is the amount of money in the present
t is the time period
r is the interest rate
V is the amount of money after a time period of t with an interest rate of r.

The investments can also be allocated in several time periods, for example years. This will define a cash flow stream, $x_0, x_1, x_2, x_3, \dots, x_n$, where x_n is the amount of money invested in year n. Each cash flow will grow interest for a different amount of years, so the future value can be calculated as:

$$FV = x_0(1 + r)^n + x_1(1 + r)^{n-1} + x_2(1 + r)^{n-2} + x_3(1 + r)^n + \cdots + x_n \tag{11}$$

7.1.1 Present value

The observation of time value of money is deeply connected with a term present value. It is obvious that value of money will grow due to interest rate. When estimating the amount of money that is needed to be invested now in order to make a certain amount of money in the future, the present value of that future amount of money is being calculated. To find out the present value of money, the future money needs to be discounted. That can be made with the help of a discount factor. Discount factor d for one year with an interest rate of r is expressed as (Luenberg 2009):

$$d = \frac{1}{(1+r)} \tag{12}$$

The present value of a cash flow stream can be also calculated by using a discount factor. In this case, each of the cash flow elements has to be discounted separately. Consequently, a different discount factor is used for each cash flow occurring in different years. The present value of a cash stream can be thought as a present payment that equals to the entire stream. The present value for a cash flow $x_0, x_1, x_2, x_3, \dots, x_n$ is calculated by (Luenberg 2009):

$$PV = x_0 + \frac{x_1}{1+r} + \frac{x_2}{(1+r)^2} + \dots + \frac{x_n}{(1+r)^n} \tag{13}$$

7.1.2 Inflation

Inflation describes the general increase in prices over the time. Due to inflation, an amount of money today has less purchasing power in the future. Inflation is measured by inflation rate. When making long-future investment decisions, it is important to observe inflation. Inflation has influence on the interest rate, and it is important to distinguish nominal interest rate from real interest rate. In nominal interest rate, inflation is not taken into account, and the real interest rate includes the effect of inflation. Money will grow interest by its nominal interest rate, but its purchasing power is deflated by inflation. Real interest rate expresses, how much an amount of money will grow in relation to its purchasing power. When nominal interest rate is r and inflation rate is f, real interest rate r_0 can be calculated as (Luenberg 2009):

$$r_0 = \frac{r-f}{1+f} \tag{14}$$

7.2 Evaluation methods of investments

When choosing between two equally good investments, the costs usually determine the investment that is chosen. There are a couple of different ways to evaluate the investment costs. Sometimes it might be difficult to compare investments, because costs can occur at different times. By evaluation methods, costs can be changed to comparable form.

7.2.1 Net present value

Net present value (NPV) is an efficient and easy way to evaluate investments. The idea is simply to calculate the present value of all cash flow streams of the investment, both positive and negative. The investment with the highest NPV can be considered as the most profitable investment. A few simple conclusions can be drawn from the value of NPV. If the NPV is positive, the investment can be held as profitable, and it is reasonable to implement it. If the NPC is zero, the investment is neither profitable nor unprofitable, and other factors must be considered when deciding whether or not implement the investment. If the NPV is negative, the investment will cause losses, and it should not be implemented by economic reasons. One strength of the NPV method is the possibility to compare two different investments, having different cash flows at different times. On the other hand, NPV method cannot be used for comparing investments that have unequal lifetimes. Net present value is calculated by the equation (13). (Luenberg 2009)

7.2.2 Annuity method

By annuity method, an investment cost is distributed equally for the investment lifetime, so that equal amount of payment is performed in an equally long period. The period can has any length, for example a year. These equal sized payments, called annuities, accumulate over time until they correspond the net present value of the investment. The size of an annuity is calculated by the means of an annuity factor, which is determined by interest rate and investment lifetime. One formula for calculating the annuity factor k is (Luenberg 2009):

$$k = \frac{r(1+r)^n}{(1+r)^n - 1} \tag{15}$$

Where r is the interest rate of one period and n is the number of periods of the annuity. When the investment cost C_i is known, annuity C_a can be calculated as:

$$C_a = k * C_i \tag{16}$$

Total annual costs C_{tot} can be calculated by adding the annual operational costs C_o to annuity:

$$C_{tot} = C_a + C_o \tag{17}$$

By annuity method, it is also possible to compare technology investments that have unequal lifetimes. Dividing the total annual costs by predicted yearly electricity output yields to costs per produced MWh_{th} for each technology. Annuity is method is applied in the cost analysis of this study.

8 Cost analysis for power plants

When investing in a new power plant, economics are usually the main principle. Electricity should be produced at the lowest possible cost. Investing in a new flue gas cleaning technology is tightly related to power plants. Thus, investments in new flue gas cleaning technologies can be calculated by using the methods for power plant cost analysis.

An interest rate should also be taken into account in power plant investments. The power plant investment may be funded by a bank loan, so the interest rate applied in the cost analysis should be higher than the interest rate included in the loan. Although the investment is funded by using own capital, the interest rate should be equal or higher than an interest rate of securities with same risk in the stock market, because otherwise the capital could be invested in these securities to provide more profit.

8.1 Basic concepts

Efficiency

In power plants, efficiency describes how much of the fuel energy content can be converted into useful form. Efficiency can be expressed as electrical efficiency (η_e) and thermal efficiency (η_{th}). Thermal efficiency is the energy content that is transferred to the steam, while the rest is wasted among the flue gas. Electrical efficiency describes the energy content of the fuel that is converted into electricity, while the rest is disposed as heat. In thermal power plants, a typical value for thermal efficiency is 90%, and 35-40% for electrical efficiency (Kaplan 2009)

Capacity of a power plant

The capacity indicates the size of a power plant. Distinction can be made between fuel capacity (MW_f), electrical capacity (MW_e) and thermal capacity (MW_{th}). Fuel capacity is the amount of fuel that can be fed into the boiler in a time unit. Electrical capacity is the electricity production in a time unit. Electrical capacity can be calculated by multiplying the thermal capacity by electrical efficiency of a power plant. Consequently, thermal capacity can be calculated by multiplying the fuel capacity by thermal efficiency. Electrical capacity can be further divided into gross capacity and net capacity. Gross capacity includes all the electrical power produced in a power plant, while net capacity is the electrical power that is fed into the electrical network. Net capacity does not include the power that is consumed by the power plant. (Kaplan 2009)

Capacity factor

Capacity factor describes a utilization rate of a power plant. It is the ratio of produced electricity during a certain time period to the amount of electricity that could have been produced if the power plant was run at full capacity for the whole period. For example, for a 100 MW power plant that produces 400 000 MWh electricity during a year, the capacity factor is (Kaplan 2009):

$$c = \frac{Amount\ of\ electricity\ porduced\ during\ a\ year}{Capacity\ of\ the\ power\ plant * 8760h} = \frac{400\ 000\ MWh}{100MW * 8760h} = 0{,}46 \qquad (18)$$

The capacity factor is rarely 1, which means that the power plant would run on its full capacity for the whole year. This is because of planned and unplanned interruptions, such as yearly maintenance procedures and unintended disturbances. Additionally, it is not always economical to feed electricity into the network. For coal-fired power plants, which usually run somewhere between baseload and peaking load plants, a typical capacity factor is somewhere between 0,25 – 0,7. (Kaplan 2009)

8.2 Cost breakdown of power plants

8.2.1 Investment costs

Power plant investments are capital intensive, which means that the initial investment costs are usually very high. Investment costs include engineering, procurement and construction costs. These are design, buying the materials and equipment, and construction of the facility. Investment costs comprise of land, building, equipment, and installation costs and overhead charges, such as transportation, stores and bookkeeping. (Raja 2006)

8.2.2 Operating costs

Operating costs include fuel costs, labour costs, maintenance and repair, storing, supervision and everything else that is needed for a normal power plant operation. Operating costs can be further divided in fixed costs and variable costs. Fixed costs are periodic costs that are independent of the produced energy. These are for example labour costs, storing, supervision and maintenance and repair. Variable costs depend on the energy output, and these are caused by electricity, water, fuel costs and any chemicals needed. (Raja 2006)

In a thermal power plant, fuel is the heaviest factor in the operating costs. In addition to actual fuel, fuel cost include also transportation, handling and storage of the fuel. Fuel costs depend on the unit price of the fuel, amount of energy produced in the power plant and the efficiency of the plant. An efficient plant requires less fuel, which decreases the fuel cost. In a steam power plant, fuel costs constitute typically 30 – 40 % of the total power generating costs. (Raja 2006)

Labour costs constitute mainly of the salaries for the staff. The amount of needed labour depends on the power plant size, complexity of the process and rate of automation. Thermal power plants are relatively complex and require more staff than hydro plants. The rate of automation is nowadays usually high, but staff is still needed to monitor the power plant operation and to perform daily maintenance and service operations. (Raja 2006)

Power plant equipment requires also periodical maintenance, such as cleaning, greasing, adjustment and overhauling, for which staff and material is needed. Maintenance costs can be estimated as an arbitrary percentage of the investment cost. Repairs are needed when something breaks down or stops. These can be minor, major and periodical and are usually charged to the depreciation fund of equipment. Again, maintenance and repair costs correlate typically with complexity of power plant process. Operation and maintenance costs of a steam power plant are typically 5-10% of the total generating costs. The cost breakdown of a power plant is shown in the Table 21. (Raja 2006)

Table 21: Cost breakdown of a power plant. (Raja 2006)

<table>
<tr><th colspan="3">Cost breakdown</th></tr>
<tr><th>Investment costs (€/kW)</th><th colspan="2">Operating costs</th></tr>
<tr><td rowspan="2">- Land
- Building
- Equipment
- Installation
- Indirect costs (Engineering, project management)
- Contingencies
- Overhead charges (Transportation, storage, bookkeeping)</td><td>Fixed costs (€/a)</td><td>Variable costs (€/MWh)</td></tr>
<tr><td>- Labour
- Maintenance and repair
- Storing</td><td>- Fuel
- Electricity
- Water
- Steam
- Waste disposal costs
- Chemicals</td></tr>
</table>

8.3 Accuracy of the cost analysis

Because power plants are designed to operate for decades, there are lot of uncertainties in the cost estimation. Variable costs are prone to price fluctuation, and especially fuel prices may change rapidly even in years. However, fuel prices tend to increase in the long run. Additionally, the mercury content of fuel can vary a lot and have a significant effect on operating costs. (Raja 2006)

Furthermore, there are a number of uncertainties regarding the installation and operating costs, such as following (Sloss 2008):

- Retrofit of the equipment may become problematic.
- Retrofit of the equipment can prolong over the scheduled plant-outage and that may cause loss of revenue.
- Balance-of-plant impacts due to system installation.

As with any new technology, costs will come down over time due to development in system efficiency and design. Only between 2004 and 2005, the costs of activated carbon injection (ACI) dropped by a factor four in the US. ACI technologies are expected to experience almost 30% reduction in capital cost by 2020, but the operating cost are expected to remain unchanged. (Sloss 2008)

Cost estimates can be made with different accuracy levels. Cost estimations have usually much uncertainty in the beginning of a project, and become more accurate towards the end of a project. The Association for the Advancement of Cost Engineering International (AACE International) uses a classification system known as 18R-97. The system has five estimate classes, 1, 2, 3, 4 and 5, where 5 is the roughest estimate and 1 is the most accurate. In the beginning of a project, the estimates fall usually in the class 5, but the accuracy level improves as the project definition becomes clearer. The classes, their accuracy ranges and their relation to the project definition

level are showed in the Table 22. Accuracy levels are shown as lower and higher boundaries, designating how much lower or higher the actual cost will be compared to the estimate. (Dysert 2003)

Table 22: Cost estimating classes according to 18R-97. (Dysert 2003)

Estimate class	Project definition (%)	Accuracy range
5	0-2	Lower: -20 to -50% Higher: 30% to 100%
4	1-15	Lower: -15 to -30% Higher level: 20 to 50%
3	10-40	Lower: -10 to -20% Higher: 10 to 30%
2	30-70	Lower: -5 to -15% Higher: 5 to 20%
1	50-100	Lower: -3 to -10% Higher: 3 to 15%

The cost estimates in this study will inevitably fall into the class 5, while no real design information is created for the analysis. The cost analysis will mostly rely on historical data that is found in the literature and the analysis will be performed by simple mathematical calculations.

8.4 Capacity-factored estimation

The nominal costs (€/kW) usually decrease as the plant capacity increases. Thus, the costs per installed capacity of a bigger plant are lower than the ones in a smaller plant. This is also known as the scales of economy. This information can be utilized when making cost estimates about new plants on the basis of existing plants. The method is called capacity-factored estimating. There is a nonlinear relationship between capacity and cost, as shown in the following equation (Dysert 2003):

$$\frac{C_A}{C_B} = \left(\frac{Cap_A}{Cap_B}\right)^e \tag{19}$$

Where,

C_A is the costs of plant A
C_B is the costs of plant B
Cap_A is the capacity of plant A
Cap_B is the capacity of plant B
e is the capacity factor

The capacity factor e describes the increment in plant costs when a smaller or larger plant is built. If the e is less than 1, the nominal costs of a plant will decrease as the plant capacity increases. Thus, a plant that is 50% larger is less than 50% more expensive. There are different values for the e, which are obtained from analysing data about realized plant costs. If there is no given value for the e, a value of 0,6 is usually used. When 0,6 is used, there is thumb rule that

doubling the capacity will increase costs by 50% and tripling the capacity will lead to 100% increase in costs. (Dysert 2003)

8.5 Units used in the cost analysis

Evaluation of power plant investments requires usually normalized units that makes it easier to compare different alternatives. Normalization means dividing the costs by some practical variable, such as power plant size (€/kW) or produced energy or electricity (€/MWh or €/kWh). In power plant investments, investment costs are usually reported in relation to power plant size (€/kW) and annual costs are usually reported in relation to produced energy or electricity (€/MWh or €/kWh). It is important to determine whether it is thermal power or electrical power that is used in the normalization.

When comparing different flue gas cleaning technologies, it is also practical to determine the cost-efficiency as the relation of removal costs to the amount of removed pollutant. The unit indicating this is €/kg of removed pollutant, such as mercury. This indicates the cost-efficiency of a removal technology. However, this method can be misused and the results are easy to misinterpret. The cost-efficiency depends much on the mercury content of a fuel and the desired removal rate. For example, if the removal rate of 90% is desired for a plant that burns fuel with a low mercury content of 0,1 ppm, the removal costs per kilogram mercury removed are higher than for a plant burning coal with a high mercury content of 1 ppm. However, more mercury in kilograms is removed in the latter case, which leads to higher operating costs. This will lead to higher costs when measured in (€/kWh).

This is why it is reasonable to report costs using different units. In this study, costs of the studied technologies are reported in three different units: increase in price of produced thermal energy (€/MWh_{th}), increase in price of produced electricity (€/kWh_e) and cost efficiency of mercury removal (€/kg Hg).

9 Cost analysis of mercury emission control technologies

9.1 Introduction to the cost analysis

Estimating the costs of mercury emission control technologies is a complex task. Behaviour of mercury at a power plant is challenging to estimate, because it depends on various factors, such as fuel composition, combustion environment and existing air pollution control technologies and their efficiencies. Mercury speciation in power plant flue gas has a strong influence on the mercury removal efficiency of air pollution control technologies, which in turn affects investment costs and operating costs. Thus, there is no one-size-fits-all solution, and it is problematic to make generalized assumptions from a cost analysis that is made for a single plant.

It should be remember that the mercury emissions depend at least on:

- mercury content of the fuel
- other chemical composition of the fuel, especially chlorine and sulphur content
- the unburnt carbon content (UBC) of the fly ash, also called loss on ignition (LOI)
- boiler type and combustion conditions
- other flue gas cleaning devices and their efficiencies
- distribution of ash between bottom ash and fly ash
- interactions between mercury and the fly ash
- temperature and velocity of the flue gas.

The purpose of this cost analysis is to estimate the costs of mercury removal that has to be implemented by power plant operators in order to comply with the emission limits of the new BREF document. The calculations are made for existing power plants that already apply some kind of flue gas cleaning devices. The emission levels are included in the BREF document as emission levels associated with best available technologies (BAT-AELs), like introduced in the chapter 5.2. Because the emission levels are expressed as lower and higher boundaries, as can be seen in the Table 8, it cannot be known what will be the exact level permitted by the national or local authority. In this analysis, the required removal efficiencies are calculated for three levels: 9, 4 and 1 µg/Nm^3, which correspond the possible highest and lowest emission levels for existing hard coal fired power plants. The main goal of the cost analysis is to explore whether these emission limits are met and if not, what the costs are caused by additional mercury removal by different technologies. The procedure for the cost analysis is described in the figure 22.

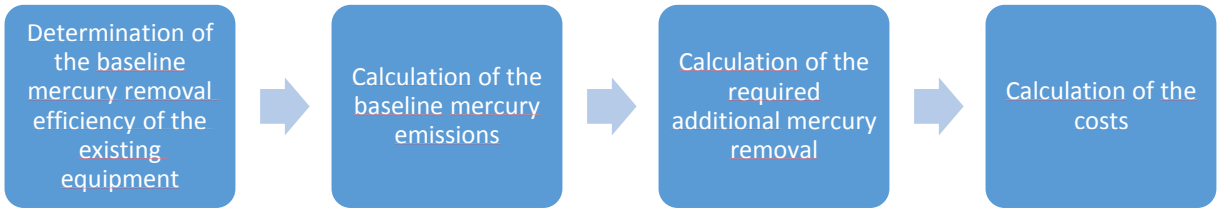

Figure 22: The calculation procedure of the cost analysis

It is reasonable to make the cost analysis for existing power plants, because most of the mercury removal installation projects are made as retrofit installations. Normally, power plants apply at

least dust control devices, and nowadays also SO_2- and NOx –control devices are common. Cost analysis is made for different flue gas cleaning configurations, which are:

1. ESP
2. FF
3. ESP + WFGD
4. ESP + SDA + FF
5. SCR + ESP + WFGD
6. SCR + ESP + SDA + FF

Where,

ESP	is electrostatic precipitator
FF	is fabric filter
WFGD	is wet flue gas desulphurization
SDA	is semi-dry absorber
SCR	is selective catalytic reduction

Although significant mercury reduction rates can be achieved by co-benefit technologies, these are not in the focus of this cost analysis. The reason is that they are mainly implemented to reduce other pollutants than mercury, such as NOx, SO_2 and dust. Thus, it is very challenging to determine the costs caused exclusively by mercury removal. It is not reasonable to allocate the costs only for mercury, because significant amounts of other pollutants are also removed. Furthermore pre-combustion technologies are also excluded. Consequently, the analysis has main emphasis in three proven and reliable mercury-specific control options, which are untreated activated carbon injection, brominated activated carbon injection and bromine injection.

9.2 Initial parameters

Coal

Because coal properties have a significant effect on mercury speciation and mercury removal efficiency, it is important to determine the coal type that is used in the analysis. The reference plant of this analysis is a coal fired power plant located in Poland, and this is why properties of typical Polish coal are used as properties for the reference coal. The parameters for the reference coal are obtained from fuel analyses that are made for coal used in Kozienice S.A. Power plant, a Polish power plant owned by an energy company called Grupa Enea. The report introduces detailed properties for four coal samples, originating from four different Polish coal mines. These mines cover about 90% of the coal burned in Kozienice. To comprise the reference coal for the cost analysis, average value of different parameters are used, except for mercury, for which the maximum value is used. (Grupa Enea 2010)

The chemical composition of the reference coal is shown in the Table 23. When compared to the coal types represented in the Table 2 in chapter 3.1., the reference coal could be described as low-sulphur bituminous coal. The chlorine content of the coal (0,200%) is high, but not unusual. In addition, the range is also high (0,037 – 0,308%). However, these are consistent with

the Table 2, according to which the chlorine content can vary between 0,004 - 0,3% in bituminous hard coals. Additionally, Hepola (2003) states that Polish coals are usually rich in chlorine, varying between 0,15 - 0,4%. It is likely that the high natural chlorine content leads to high rates of oxidized mercury in the flue gas, which facilitates essentially the mercury removal. Moreover, the average sulphur content (0,78 %) is low, which may also be beneficial for mercury removal. The average mercury content (0,139 ppm) seems to be consistent with the results of the study made by Lorenz, Grudziński (2008) that was introduced in the chapter 3.1. In that study, an average mercury content of 0,1 – 0,15 ppm for Polish hard coal was reported.

Table 23: Parameters of the reference coal used in the analysis and the parameter ranges of the analysed coal samples. (Grupa Enea 2010)

	Reference coal (Average values, *maximum value)	Range
Gross calorific value (kJ/kg)	23 850	22 467 - 25231
Net calorific value (kJ/kg)	22 803	21 392 – 24 162
Moisture (%)	9,60	8,7 – 10,9
Ash (%)	19,28	16 - 23
Carbon, C (%)	58,51	54,66 – 61,84
Hydrogen, H (%)	3,70	3,63 – 3,85
Nitrogen, N (%)	0,96	0,84 – 1,08
Oxygen, O (%)	6,99	5,61 – 8,25
Sulphur, S (%)	0,78	0,36 – 1,03
Chlorine, Cl (%)	0,200	0,037 – 0,308
Mercury (ppm)	0,139*	0,048 – 0,139

Power plant parameters

Power plant parameters used in the cost analysis are represented in the Table 24. Calculations are made for three pulverized coal fired power plants with different capacities. The electrical capacities are 100, 250 and 500 MW_e, With an electrical efficiency of 38%, the corresponding thermal capacities are 263, 658 and 1316 MW_{th}. Operating hours are estimated to be 6000h in a year, which gives a capacity factor of 0,68. Loss on ignition is 1%, which means that 1% of the fly ash consists of carbon that has left the boiler as unburnt.

Table 24: Power plant parameters used in the cost analysis.

Electrical capacity	100 MW_e, 250 MW_e, 500 MW_e
Thermal capacity	263 MW_{th}, 658 MW_{th}, 1316 MW_{th}
Furnace type	Pulverized coal burner
Operating hours	6000 h/a
Capacity factor	0,68
Electrical efficiency	38%
Operating load	100 %
Loss on ignition (LOI)	1 % (of fly ash)
Share of bottom ash from total ash	25 %

Flue gas analysis

The allowed mercury emission levels associated with the best available techniques (BAT-AELs) are expressed as micrograms per normal cubic meters (µg/Nm^3) in the following conditions: dry flue gas at a temperature of 273,15 K (0 °C) and a pressure of 101,3 kPa. Additionally, the BAT-AELs are reported in a reference oxygen level of 6%. To determine the initial mercury emissions in these conditions and units, the composition and volume streams of the flue gases were calculated.

Some basic assumptions were made in the flue gas analysis. Minor elements of the fuel, such as sulphur, nitrogen and chlorine were excluded from the calculations. Furthermore, the possible water content of combustion air was also excluded. The combustion process was estimated to be perfect, and gases like nitrogen and argon were not expected to react with oxygen. A combustion air coefficient of 1,4 was used to determine the required combustion air. Additionally, all of the mercury in the fuel was expected to exit the boiler along with the flue gas. In reality, a minor fraction (< 1%) is deposited in the bottom ash.

Because mercury BAT-AELs are measured in dry flue gas, water was finally excluded from the flue gas stream. Mercury emissions were correlated in 6% oxygen level by using the formula (20), as given in the BAT-BREF document (European IPPC Bureau 2016):

$$E_R = \frac{21-O_R}{21-O_M} * E_M \quad (20)$$

Where

E_R is the emission concentration at the reference oxygen level O_R
O_R is the reference oxygen level in vol-%
E_M is the measured emission concentration
O_M is the measured oxygen concentration

The main results of the flue gas analysis are shown in the Table 25. The main components of the dry flue gas are nitrogen (N_2), oxygen (O_2), and carbon dioxide (CO_2). The flue gas volume stream was calculated by using the molar volume an ideal gas at 1 atmosphere and 0 °C, which is 22,41 dm^3/mol. The pressure and temperature correspond the standard conditions applied in the BAT-AELs. The main components of the flue gas are nitrogen (N_2), oxygen (O_2) and carbon dioxide (CO_2). As a result, a mercury input flow of 17,14 µg/Nm^3 (6% O_2, dry) was calculated for each plant. This would also be the stack emission level of the plants, if no mercury removal was occurred. With the input flow of 17,14 µg/ Nm^3, minimum total removal rates of 47,5%, 76,7% and 94,2% would be required for the emission levels 9, 4 and 1 µg/Nm^3, respectively.

Table 25: The main results of the flue gas analysis for different plant sizes.

	263 MW_{th}	658 MW_{th}	1316 MW_{th}
Flue gas composition, dry			
- Carbon dioxide, CO_2 (vol-%)	12,8	12,8	12,8
- Nitrogen, N_2 (vol-%)	80,5	80,5	80,5
- Oxygen, O_2 (vol-%)	6,7	6,7	6,7
Flue gas volume stream (Nm^3/h)	353 000	884 000	1 767 000
Hg input (g/h)	5,28	14,44	28,88
Hg input (µg/Nm^3) (6% O_2, dry)	17,14	17,14	17,14
Upper Hg emission level (µg/Nm^3) (6% O_2, dry)	9	4	4
Lower Hg emission level (µg/Nm^3) (6% O_2, dry)	1	1	1

9.3 Baseline mercury removal and baseline emissions

The cost analysis is made for power plants that already have some kind of air pollution control devices. Because these devices remove also mercury as a co-benefit effect, it is necessary to determine a baseline mercury removal rate that already takes place in these plants. Determining the initial removal level of different air pollution control configurations is not an easy task. As mentioned before, there are many factors that have an effect on the total removal efficiency of a plant.

This study will rely on the mercury removal rates that have been introduced in different literature sources. These are based on various laboratory or plant tests made with different configurations. The Table 20 in the chapter 6.4 summarized the removal rates of different configurations and coal types. The average values of these removal rates are used as baseline mercury removal rates in this cost analysis. Because the coal used in this cost analysis corresponds bituminous coal, the values for this coal type are used. As a removal efficiency for an ESP, a cold-side ESP will be used, because this corresponds a typical configuration of ESP devices. The baseline mercury removal efficiencies of the configurations included in the cost analysis are shown in the Figure 23.

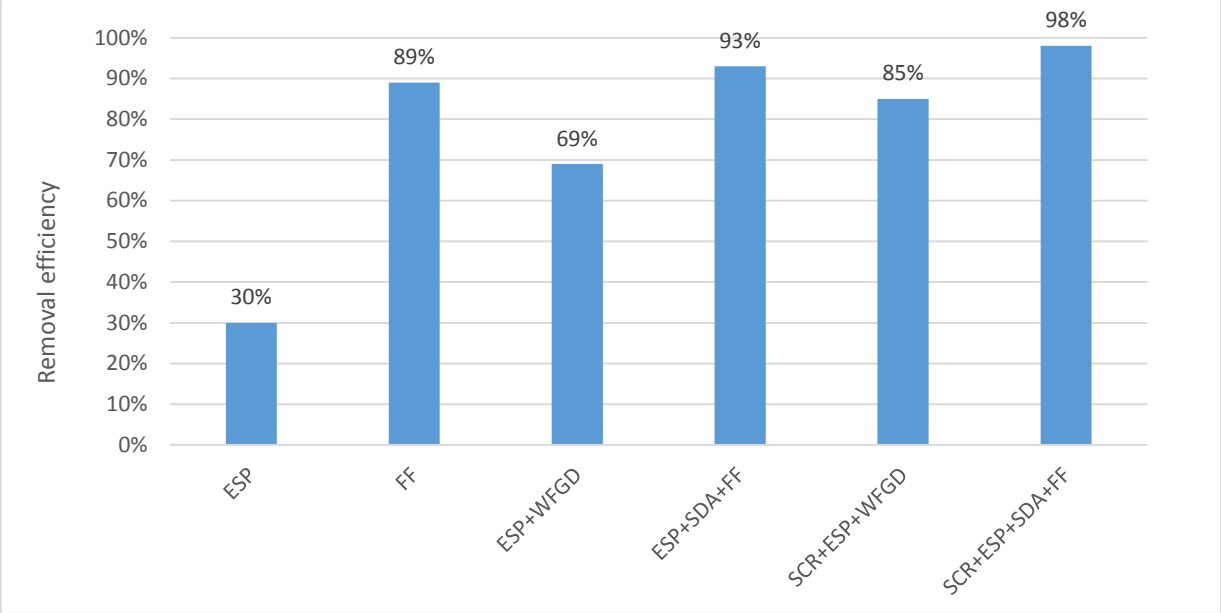

Figure 23: Baseline removal efficiency of different flue gas cleaning configurations used in the cost analysis.

The removal efficiencies of different configurations seem to have a reasonable rank order. As can be seen in the Figure 23, high removal rates are already achieved by the existing flue gas cleaning devices. This can be explained by the reference coal type, which is bituminous high chlorine coal. This likely results in high shares of oxidized and particle bound mercury that are easy to capture. There is always much uncertainty when estimating removal efficiencies without knowing the plant-specific conditions. However, when comparing the removal rates of the Figure 23 to the ones for bituminous coal in the Figure 21, it can be seen that they are quite similar to each other. The data for the Figure 21 is obtained by US EPA from field tests in plants with different flue gas cleaning configurations. Thus, the baseline removal efficiencies used in this cost analysis can be regarded as good estimations.

By using the baseline mercury removal efficiencies of the Figure 23, the baseline mercury emissions of the examined plants were calculated. The detailed calculations can be seen in the Appendices 2,3 and 4. Mercury emissions can be expressed in two different units: mass per time (g/h) or (kg/a) and mass per flue gas volume (µg/Nm^3). The removal efficiency is estimated to be the same regardless of the thermal capacity of the boiler. Because larger power plants produce more electricity in the given time unit, they also emit more mercury within the same time period (g/h) or (kg/a). However the concentration of mercury in the flue gas remains the same. In other words, when mercury emissions are expressed as µg/Nm^3, they are the same in all plant sizes. Because the allowed emission levels of the Industrial Emission Directive are expressed as µg/Nm^3, that unit is more practical. The mercury emissions after each flue gas cleaning configuration and their relation to allowed emission limits are showed in the Figure 24.

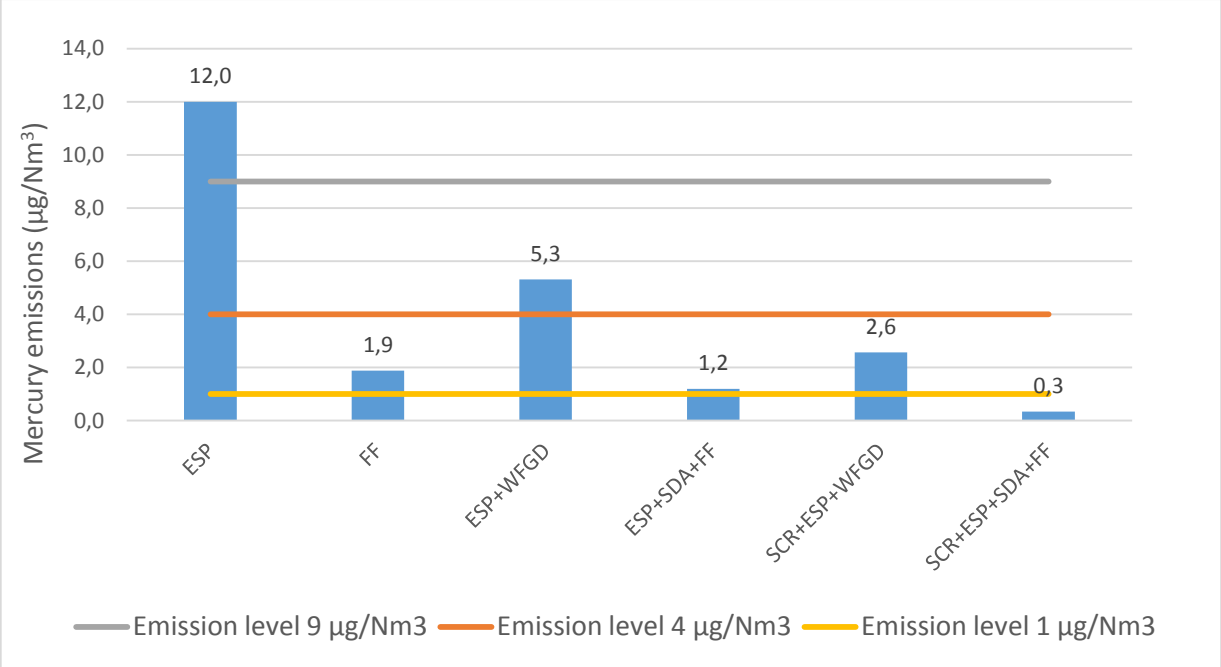

Figure 24: Mercury air emissions in different flue gas cleaning configurations and allowed emission limits.

As can be seen in the Figure 24, all the other configurations except ESP emit less than 9 µg/Nm^3, which is the upper limit for small plants (<300 MW_{th}). Besides ESP, plants with ESP + WFGD emit more than 4 µg/Nm^3, which is the upper limit for large plants (≥300 MW_{th}). When the lowest limit 1 µg/Nm^3 is applied, only plants with a configuration SCR+ESP+SDA+FF will achieve that limit. This means that all other configurations need to implement additional mercury removal technology, when the emission limit of 1 µg/Nm^3 is applied.

By this basis, the required additional removal rate was calculated. The total removal rate can be thought to be consisting of the removal rate of existing equipment and removal rate of the additional mercury removal equipment. In principal, the total removal can be calculated by the formula (Yelverton 2009):

$$f_{total} = 1 - \left[\left(1 - f_{existing\ equipment}\right) * \left(1 - f_{additional\ equipment}\right)\right] \quad (21)$$

Where,

f_{total} is the total removal rate
$f_{existing\ equipment}$ is the removal rate of existing equipment
$f_{additional\ equipment}$ is the removal rate of additional equipment

Therefore, the required additional removal rate can be calculated as:

$$f_{additional\ equipment} = 1 - \frac{1 - f_{total}}{1 - f_{existing\ equipment}} \quad (22)$$

The results are showed in the Table 26. The Table tells how much mercury is needed to be removed after each configuration, when some of the emission levels 9, 4 or 1 µg/Nm^3 have to be achieved. As can be seen from the Table 26, SCR+ESP+SDA+FF achieves even the tightest emission level 1 µg/Nm^3 and no additional measures have to be taken. The mass flows for all studied flue gas cleaning configurations with plant size 263 MW_{th} are illustrated in the appendix 1.

Table 26: Additional removal requirements for flue gas cleaning configurations for different emission limits.

APCD	Emission limit 9µ/Nm^3	Emission limit 4µ/Nm^3	Emission limit 1µ/Nm^3
ESP	25,0 % 3,0 µg/Nm^3	66,7 % 8,0 µg/Nm^3	91,7 % 11,0 µg/Nm^3
FF	0,0 % 0,0 µg/Nm^3	0,0 % 0,0 µg/Nm^3	47,0 % 0,9 µg/Nm^3
ESP+WFGD	0,0 % 0,0 µg/Nm^3	24,7 % 1,3 µg/Nm^3	81,2 % 4,3 µg/Nm^3
ESP+SDA+FF	0,0 % 0,0 µg/Nm^3	0,0 % 0,0 µg/Nm^3	16,6 % 0,2 µg/Nm^3
SCR+ESP+WFGD	0,0 % 0,0 µg/Nm^3	0,0 % 0,0 µg/Nm^3	61,1 % 1,6 µg/Nm^3
SCR+ESP+SDA+FF	0,0 % 0,0 µg/Nm^3	0,0 % 0,0 µg/Nm^3	0,0 % 0,0 µg/Nm^3

9.4 Cost analysis

The costs of mercury control have been calculated for three different mercury-specific removal technologies, which are activated carbon injection, brominated activated carbon injection and bromine injection. The detailed calculations for these technologies are shown in the appendices 2, 3 and 4, respectively. The costs have been estimated mostly by values found in literature, and they are usually based on experiences in realized projects. The costs are calculated and represented for cases that require additional mercury removal. This is why SCR+ESP+SDA+FF is excluded from the analysis, because this configuration can achieve even the lowest emission limit of 1 µg/Nm^3 without any additional removal technology.

The investment costs are assumed to include major and auxiliary equipment, installation, project management and engineering, freight, taxes, start-up cost and other capitalized costs. The annualized costs have been calculated for 15 years plant lifetime with a real interest rate of 8 %, which includes also the effect of inflation.

In the BREF document, the measuring requirement for mercury emissions is described as “yearly average or average of samples obtained during one year”, as described in the chapter 5.2. Thus, it assumed that no continuous emission measurement system (CEMS) is required for the mercury-specific installations.

9.4.1 Cost and consumption values used in the analysis

Values used in operating cost

The operating costs are represented in the Table 27. Cost for activated carbon and brominated activated carbon are obtained from literature. Electricity price is the average electricity price for industrial customers in Poland during the second half of 2015. It is assumed that the increase of operating costs equals the general inflation rate, so the increase in the costs is balanced by the inflation. Thus the general cost increase is excluded from the calculations. Many of the prices have been reported in U.S. dollars in the original source. Conversions to euros has been made by using a conversion rate of 1 $ = 0,89 €.

Table 27: Values used in the operating costs.

Factor	Cost
Untreated activated carbon	1,33 €/kg[1)]
Brominated activated carbon	1,78 €/kg[1)]
Bromine salt solution ($CaBr_2$ 52 w%)	1,96 €/kg[3)]
Electricity	81,3 €/MWh[2)]
Waste disposal costs	70 €/ton[4)]
Fly ash revenue	0 €/ton[4)]
Sources: 1) (Reissner, Crèvecoeur et al. 2015) 2) (Eurostat 2016) 3) (Leonard et al. 2011) 4) (Leino, 2016).	

Activated carbon consumption

The consumption rate of activated carbon is difficult to estimate, because there is not a simple chemical reaction formula for the adsorption. Adsorption consists of complex chemical and physical reactions that depend on many factors, as described in the chapter 6.3.1. In real power plant projects, exact consumption rates are determined by tests and measurements. Thus, the values vary with fuel and power plant characteristics. However U.S. Environmental Protection Agency (EPA) has developed an algorithm, which is based on field test in power plants with different coal types and particulate matter control devices (electrostatic precipitator or fabric filter). This algorithm is applied in this study to estimate the activated carbon injection rates in different situations.

The algorithm is following (Yelverton 2009):

$$y = \log(injection\ rate) = Ax^2 + Bx + C \quad (23)$$

where x is the desired removal rate as %
y is the logarithmic function of the activated carbon injection rate as lb/MMafc (pounds per million actual cubic feets)
A, B and C are specific constants that are provided in the Table 28.

Table 28: Constants for the equation (23). (Yelverton 2009)

	A	B	C
PAC, Bituminous FF	1,6944	-1,1267	-0,0009
PAC, Bituminous ESP	-0,6647	2,1232	-0,0665
Treated PAC Subbituminous FF	0	2,5007	-2,2097
Treated PAC, Bituminous ESP	0	1,207	-0,2277

Consumption rates for all the four situations of the Table 28 were calculated. Because there was no constants for treated PAC in plants combusting bituminous coal and applying fabric filter, the constants for subbituminous coal were used. The results for the injection rates of the activated carbon as a function of desired mercury removal rates are shown in the Figure 25. Because the activated carbon consumption rate in the equation (23) is initially expressed as lb/MMafc, it was converted to mg/Nm^3. The conversion rate is approximately 1 lb/MMacf = 16,02 mg/ Nm^3.

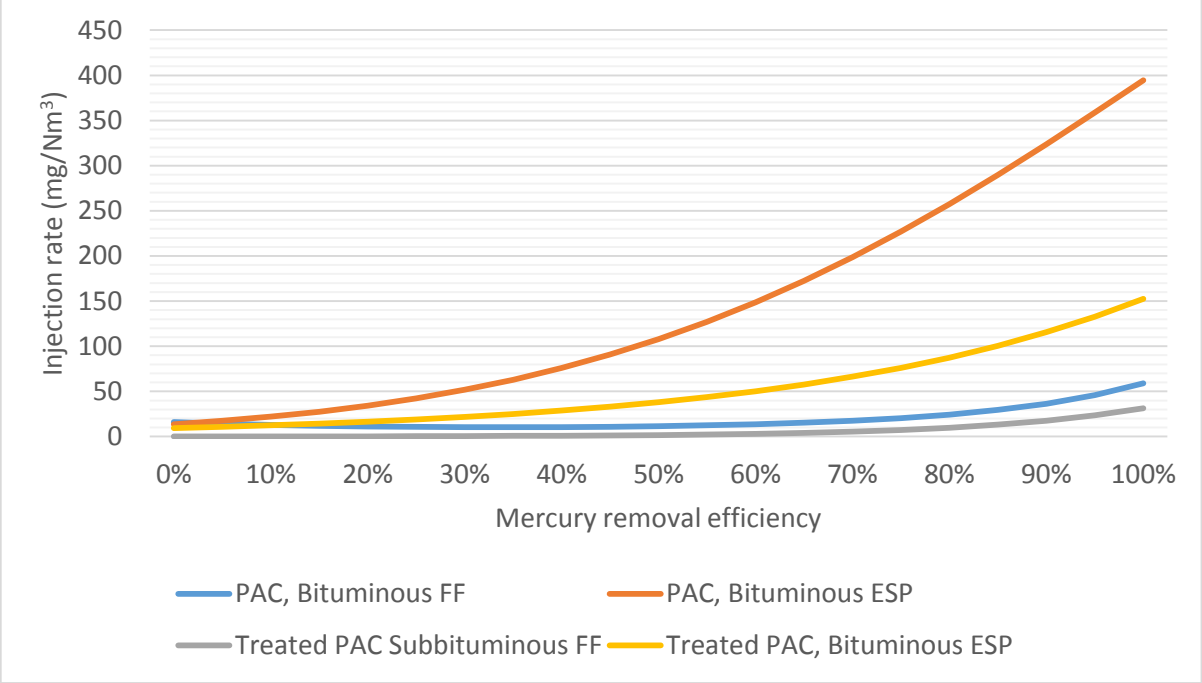

Figure 25: Activated carbon injection rate as a function of targeted mercury removal rate.

As can be seen from the Figure 25, much less activated carbon is required when a fabric filter is used. According to the algorithm, an injection rate of 320 mg/Nm^3 is required for a power plant with an ESP, when 90% removal rate is desired. In a plant with a FF, an activated carbon injection rate of 35 mg/Nm^3 is sufficient to achieve the same removal rate. As a reference, in a report made by Reissner, Crèvecoeur et al. 2015, activated carbon consumption rate of 80-200mg/Nm^3 is reported for an ESP and 20-50mg/Nm^3 for a FF, when 90% removal rate is desired.

To evaluate the results calculated by the algorithm, comparisons to literature data were made. Data is based on field test measurements on real power plants. The algorithm estimation for plants with an ESP and its relation to field test data is shown in the Figure 26. The detailed information of the data points is represented in the Table 29. The Figure shows the data for both untreated and halogenated activated carbon. As can be seen, the field data point set mostly below to the estimation lines. This implicates that in reality, not that much activated carbon is needed than the algorithm estimates. The algorithm estimates that activated carbon consumption approaches zero, when the removal rate is decreased. In this study, it is estimated that the minimum activated carbon injection rate is 5mg/Nm^3.

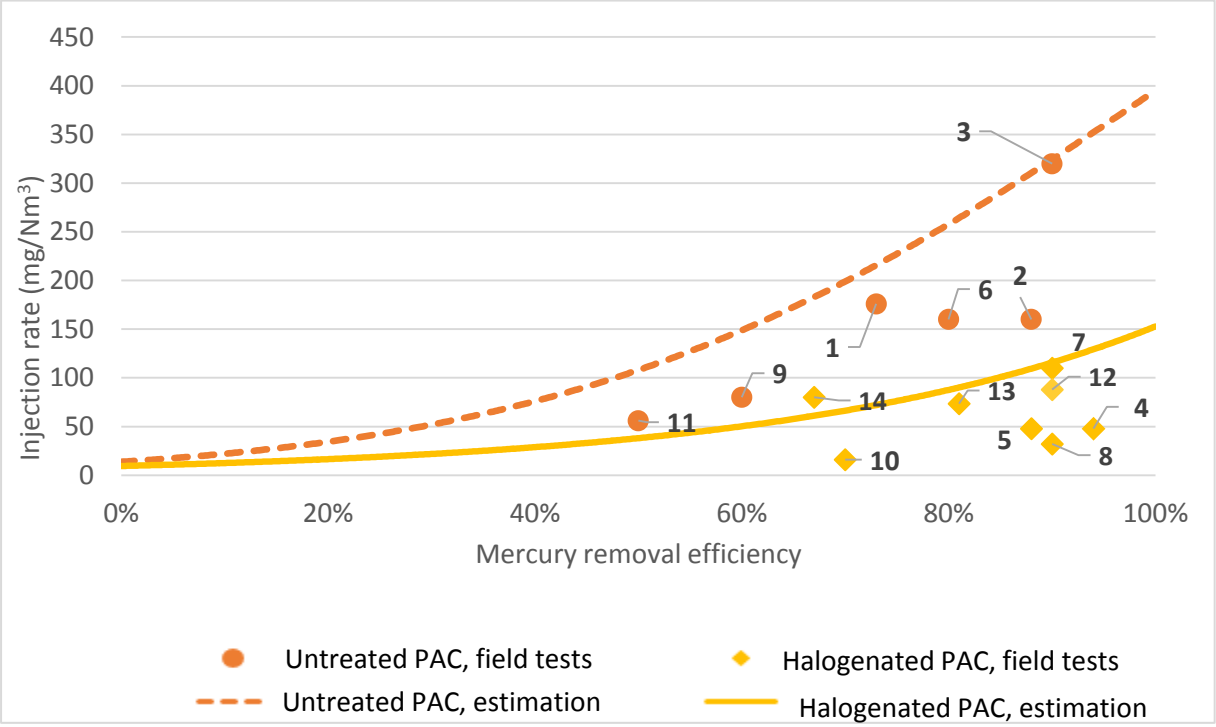

Figure 26: Activated carbon injection rate as a function of targeted mercury removal rate in plants with an ESP.

Table 29: Field test data related to the Figure 26.

	Plant	APCD	Coal	Source
1	Pleasant Prairie, USA	CS-ESP	Subbituminous	(Srivastava, Hutson et al. 2006)
2	Salem Harbor, USA	CS-ESP	Low-S Bituminous	(Srivastava, Hutson et al. 2006)
3	Brayton, USA	2 x CS-ESP	Low-S Bituminous	(Srivastava, Hutson et al. 2006)
4	Meramac, USA	CS-ESP	Low-S Bituminous	(Srivastava, Hutson et al. 2006)
5	St. Clair, USA	CS-ESP	Subbituminous	(Srivastava, Hutson et al. 2006)
6	Cherepetskaya, Russia	CS-ESP	Lignite	(Zykov, Anichkov et al. 2014)
7	Cherepetskaya, Russia	CS-ESP	Lignite	(Zykov, Anichkov et al. 2014)
8	Leland Olds Station, USA	ESP	Lignite	(Feeley III, Jones et al. 2009)
9	Leland Olds Station, USA	ESP	Lignite	(Feeley III, Jones et al. 2009)
10	Stanton Station, USA	ESP	Subbituminous	(Feeley III, Jones et al. 2009)
11	Stanton Station, USA	ESP	Subbituminous	(Feeley III, Jones et al. 2009)
12	Presque Isle, USA	TOXECON II	Subbituminous	(Feeley III, Jones et al. 2009)
13	Crawford Station, USA	ESP	Subbituminous	(Feeley III, Jones et al. 2009)
14	Will County, USA	HS-ESP	Subbituminous	(Feeley III, Jones et al. 2009)

The corresponding information for plants with a FF are shown in the Figure 27 and in the Table 30. Fewer field test data were found in the literature for plants with a FF. However, the field test data points for halogenated activated carbon are settled in the same line with the estimation curve. This support the results given by the algorithm. The data points for untreated activated carbon deviate more from the estimation curve. The Figure 27 shows also the strange shape of estimation curve for untreated activated carbon. At small removal rates, activated carbon consumption seems to increase when removal rate decreases. Thus, it could be said that the values given by the algorithm are not very reliable at small removal rates. Also with this case, the minimum injection rate is set to 5 mg/Nm^3.

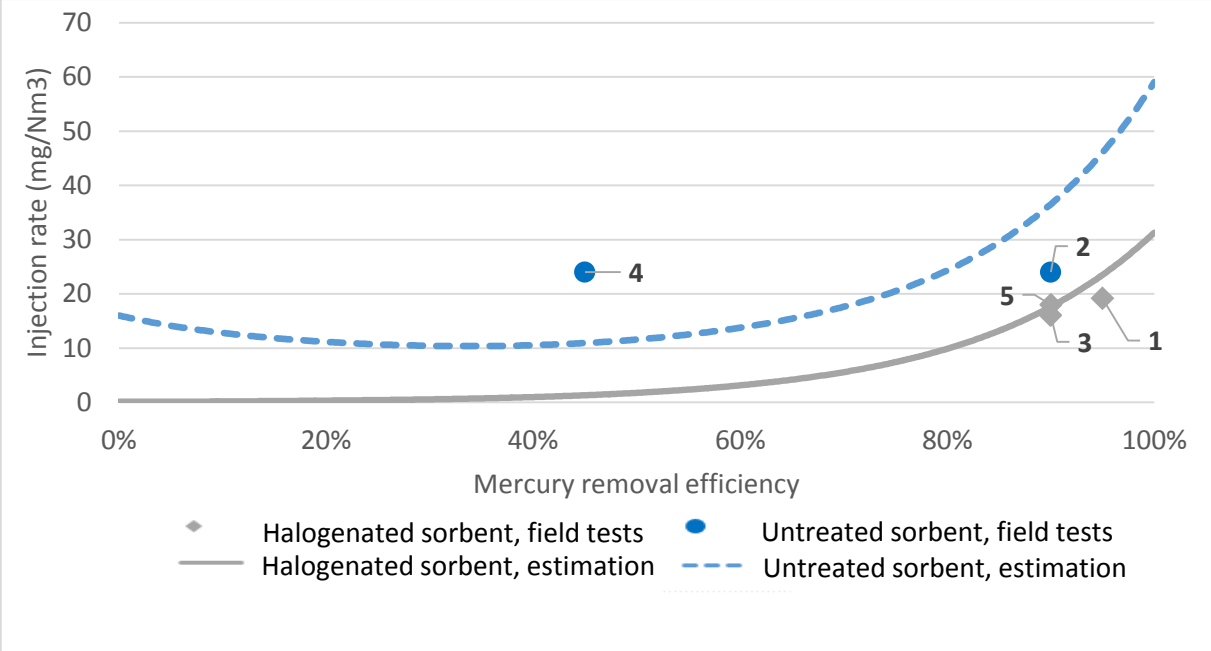

Figure 27: Activated carbon injection rate as a function of targeted mercury removal rate in plants with a FF.

Table 30: Field test data related to the Figure 27.

	Plant	APCD	Coal	Source
1	Holcomb, USA	SDA + FF	Subbituminous	(Srivastava, Hutson et al. 2006)
2	Gaston, USA	TOXECON	Low-S Bituminous	(Srivastava, Hutson et al. 2006)
3	Stanton Station, USA	FF	Lignite	(Feeley III, Jones et al. 2009)
4	Stanton Station, USA	FF	Lignite	(Feeley III, Jones et al. 2009)
5	Presque Island, USA	TOXECON	Subbituminous	(Feeley III, Jones et al. 2009)

9.4.2 Untreated activated carbon injection

Investment costs

The investment costs of an activated carbon injection system are based on the literature sources. The investment costs consist of design and installation of the following process equipment (Granite, Pennline et al. 2014):

- storage silo
- screw feeders (conveyors)
- metering system
- injection system (including metering, blowers, manifolds and lances).

Cost of the storage silo depends mainly on size. The size has to be adapted to the unique needs of the plant, such as boiler capacity and the mercury removal rate that has to be achieved. Higher power plant capacity and removal rate mean higher activated carbon consumption, which in turn requires a larger storage silo. Because the activated carbon is usually delivered by a truck, the

minimum size of the silo is typically larger than the truck capacity. A smaller silo would mean that the silo should be emptied completely before refill, which would mean interruptions in activated carbon delivery. (Granite, Pennline et al. 2014)

Metering is needed for monitoring the amount of activated carbon that is fed in the flue gas. It is important that accurate amount of activated carbon is injected, because unnecessarily high consumption means extra costs. On the other hand, too less activated carbon means that the target reduction rate will not be met. There are mainly two types of meters, gravimetric and volumetric, from which gravimetric meters are more accurate but cost two or three times more. In addition, volumetric meters have to be recalibrated each time when different activated carbon type is used. (Granite, Pennline et al. 2014)

The conveying system consist of piping and blowers and motors that are needed to overcome the pressure drop in the pipelines. The costs of the conveying system depend on conveying distance, elbows, amount of activated carbon fed and the change in the elevation. Blowers have to be designed such a way that sufficient velocity is provided in the pipelines. Too low velocity will result in plugging and fallout of the activated carbon, and two high velocity may damage the downstream equipment. (Granite, Pennline et al. 2014)

Injection system comprise the pipework, manifolds and lances. The larger and more complex the plant is, more costly the injection system will be. It is important that the manifolds and lances are designed to meet the accurate amount of activated carbon that is needed to be injected in the flue gas. (Granite, Pennline et al. 2014)

It is challenging to find information about the investment costs of realized activated carbon injection systems. Because most of the experiments on mercury removal technologies have taken place in the United States, the cost information is usually based on realized plants in the US. According to Lipinski, Leonard et al. (2011), the capital costs of an activated carbon injection system vary between 1,78 – 8,90 €/kW_e (2-10 \$/kW_e), depending on the unit size. Sloss (2008) has also made a wide study on costs of mercury removal technologies. According to that, the capital costs for an activated carbon injection system are approximately following depending on the plant size:

- 5,34 – 8,01 €/kW_e (6 – 9 \$/kW_e) for plant sizes <200 MW_e
- 3,56 - 5,34 €/kW_e (4 – 6 \$/kW_e) for plant sizes 200 – 360 MW_e
- 2,67 – 3,56 €/kW_e (3 – 4 \$/kW_e) for plant sizes >360 MW_e.

The upper values of the investment costs represented above are used in this study. The installation costs for the plant sizes 100 MW_e, 250 MW_e and 500 MW_e are 8,01 €/kW_e, 5,34 €/kW_e and 3,56 €/kW_e, respectively. The costs should be held as approximate, because they do not include the detailed design and retrofit assessment. Additional costs may occur due to ductwork-modifications and foundation installations. The investment costs of an activated carbon injection system are demonstrated in the Figure 28.

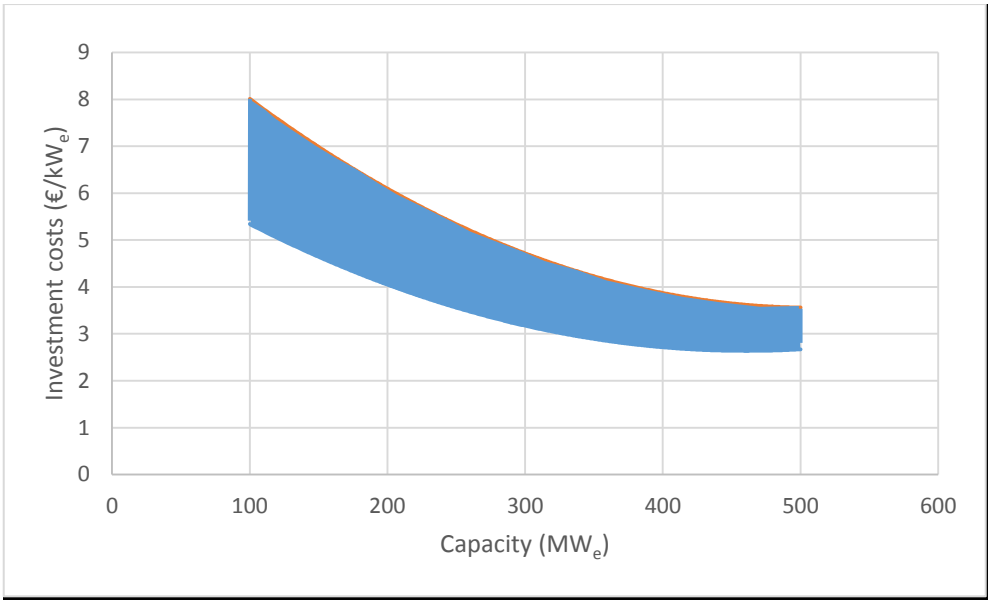

Figure 28: Investment costs of an activated carbon injection system. Data source: (Sloss 2008).

Operating costs

The operating and maintenance costs of an activated carbon injection system are (Sloss 2008)

- variable costs:
  - activated carbon
  - activated carbon disposal, if any
  - electricity
- fixed costs:
  - maintenance

The variable costs of an activated carbon injection system consist of consumption of the activated carbon itself and electricity. Also, the used activated carbon may be hazardous waste that has to be disposed, which causes costs. Activated carbon cost is by far the biggest cost component of operating costs, comprising about 95% of all operating costs. Electricity is required for the motors of the screw feeder system and for the blowers of the injection system. (Sloss 2008)

Critical factors to the sorbent price are distance between activated carbon supply location and power plant location and the quantity of activated carbon that is traded. Because there are many activated carbon suppliers, the cost is not fixed and will fluctuate with market forces. However, suppliers may offer a discount price to power plant operators over a fixed period. It is likely, that activated carbon costs decline with competition and technological maturation. According to ICAC (Institute of Clean Air Companies), cost of activated carbon injection dropped fourfold in 2004-2005. However, as more stringent legislation for mercury emissions will be seen in many countries, the demand for activated carbon will grow, which in turn may lead to higher prices. (Sloss 2008)

There are a lot of activated carbon suppliers with various carbon types for mercury control. Prices can be found in literature, but because of aforementioned factors, the prices may become outdated quickly. Sloss (2008) has listed some activated carbon prices and the prices vary between 0,77 – 2,67 €/kg (0,87 – 3 $/kg) depending on the supplier and the activated carbon type. Untreated activated carbon is usually cheaper, and the prices for these are 0,77 – 1,07 €/kg (0,87 -1,2 $/kg). This study will rely on the more recent information of the report provided by Reissner, Crèvecoeur et al. (2015), where a price of 1,33 €/kg was presented for untreated activated carbon.

The main electricity consumers are the blowers and the screw feeders that are needed to convey the activated carbon into the flue gas. The electricity consumption of the screw feeders is estimated to be negligible, so it is excluded from the calculation. The electricity consumption of the blowers depends mainly on the air volume flow and pressure. In this study, it is estimated that an activated carbon/air ratio of 1 kg/m^3 is sufficient to provide proper distribution of activated carbon into the flue gas. Because the activated carbon consumption varies according to the targeted removal rate, blowers are dimensioned such a way that they can inject 500mg/Nm^3 of activated carbon. This will be more than enough for the highest removal rates in this study. The blower air volume is thought to constant, regardless of the activated carbon injection rate. With the injection rate of 500mg/Nm^3, the blower air flows for the plants 263 MW$_{th}$, 658 MW$_{th}$ and 1316 MW$_{th}$ are 184 m^3/h, 459 m^3/h and 918 m^3/h, respectively.

The power consumption of the blowers were calculated by the equation (24) (Wiksten 2009). It is estimated that the blower efficiency is 0,7 and motor efficiency is 0,9. The pressure drop in the system is assumed to be 80 kPa, as in the reference plant. As a result, the power consumption of the blowers for the plant sizes 263 MW$_{th}$, 658 MW$_{th}$ and 1316 MW$_{th}$ are 6,5 kW, 16,2 kW and 32,4 kW, respectively. As can be seen, the required electricity consumption is low.

$$P = \frac{\dot{V} \Delta p_{tot}}{\eta_m \eta_e} \tag{24}$$

Where,

P	is power (kW)
$\dot{V}$	is air volume flow (Nm^3/s)
Δp_{tot}	is total pressure drop (kPa)
η_m	is blower efficiency
η_e	is motor efficiency

As mentioned before, activated carbon may have negative effects on the fly ash quality and make it unmarketable. This is due to fact that that too high carbon content in the fly ash decreases its quality and applicability in cement production. In general, the fly ash quality is seen to be affected, if the unburned carbon content is more than 5%. Because not much data is available on fly ash price and disposal costs, this study will rely on information acquired in Finland. According to Leino (2016), the price of fly ash in Finland fluctuates both sides of 0 €/kg, and

can be even negative. If fly ash is unusable and it has to be disposed, the disposal costs in Finland are around 70 €/ton. (Leino, 2016)

In this study, the amount of fly ash for each plant has been calculated by ash contents of 19,3%, as shown in the Table 23. Fly ash is estimated to comprise 75 % of the total ash. The initial unburned carbon, also loss on ignition (LOI) content of the fly ash is assumed to be 1%, as seen in the Table 24. It was calculated, that the amount of activated carbon that would exceed the critical 5 % is 650 mg/Nm^3. In this study, the highest required injection rate is 336 mg/Nm³, which is for plants with an ESP only, when emission level 1 µg/Nm³ is desired. With that injection rate, the unburned carbon content remains less than 5%. As a conclusion, additional fly ash disposal costs are not occurred due to activated carbon injection and they can be excluded from the calculations.

Regardless of the high automation rate, some staff is required for monitoring the normal operation of the plant, reacting abnormal situations and performing daily maintenance procedures. The staff needed for the ACI system consists of the operating staff of the plant. In this study, it is estimated that the power plant staff is sufficient to take care of the ACI system, so no additional staff is needed. Thus, the staff costs can be excluded.

The consumption values of operating costs are shown in the Table 31. The consumption of activated carbon is calculated by the equation (23), and it expresses the injection rate that is needed for each flue gas cleaning configuration to achieve the required emission level. The equation (23) has different constants for plants with an ESP and plants with a FF, implicating that plants with a FF are more efficient to remove mercury. The relation between activated carbon consumption and required removal rate is also shown in the Figure 25, with the exception that the minimum injection rate in this study is regarded as 5 mg/Nm³. All the studied plant configurations in this cost analysis have either an ESP or a FF, and the corresponding constants are applied for them. It is assumed that other air pollution control devices do not have influence on the activated carbon consumption.

Table 31: Consumption values for operating costs of activated carbon injection.

	263 MW_{th}			658 MW_{th}			1316 MW_{th}		
Emission level (µg/Nm³)	9	4	1	9	4	1	9	4	1
Activated carbon consumption (kg/h)									
- ESP	15,6	66,5	123,2	38,9	166,2	308,0	77,8	332,5	616,0
- FF	-	-	4,1		-	10,3		-	20,6
- ESP+WFGD	-	15,4	97,4		38,4	243,4		76,9	486,8
- ESP+SDA+FF	-	-	4,3			10,6			21,3
- SCR+ESP+WFGD	-	-	56,5			141,3			282,6
- SCR+ESP+SDA+FF		-	-			-			-
Electricity consumption (kW)	6,5			16,2			32,4		
Maintenance (%/investment costs)	3			3			3		

Total costs

The total costs for plant sizes 263 – 1316 MW_{th} are summarized in in the Table 32. The Table shows the cost range for different existing air pollution control configurations and for different target emission levels. It also shows the required additional removal rate that has to be reached by activated carbon injection. The cases that do not require additional mercury removal are excluded for the Table 32. As can be seen, the annual costs and increase in electricity price is higher, when the additional removal rate is increased. This is because of the increased activated carbon consumption. On the contrary, the cost efficiency (€/kg Hg removed) may be very high, although the other cost indicators have low values. This is due to very low mercury removal rates in some cases, which means that the amount of removed mercury in relation to costs is very low.

Table 32: Summary of the total costs of untreated activated carbon injection for different emission levels.

Existing air pollution control configuration	Required additional removal (%)	Annual costs (€/MWh_{th})	Increase in electricity price (cent/kWh_e)	Cost efficiency (€/kg Hg)
9µg/Nm^3				
- ESP	25	0,113 – 0,153	0,030 – 0,040	29 400 – 39 800
4µg/Nm^3				
- ESP	66,7	0,370 – 0,410	0,097 – 0,108	36 100 – 40 100
- ESP+WFGD	24,7	0,112 – 0,152	0,029 – 0,040	66 500 – 90 400
1µg/Nm^3				
- ESP	91,7	0,657 – 0,697	0,173 – 0,183	46 600 – 49 500
- FF	47,0	0,055 – 0,095	0,014 – 0,025	48 400 – 83 800
- ESP+WFGD	81,2	0,526 – 0,566	0,138 – 0,149	95 200 – 102 400
- ESP+SDA+FF	16,6	0,056 – 0,096	0,015 – 0,025	217 500 – 374 400
- SCR+ESP+WFGD	61,1	0,320– 0,360	0,084 – 0,095	159 000 – 178 900

The annual costs and cost efficiency of activated carbon injection for different air pollution configurations and plant sizes are illustrated in the Figure 29. The applied emission level is 1 µg/Nm^3. As can be seen, the usual trend is that the costs per produced MWh_{th} or removed kilogram of mercury decrease when the plant size increases. In the case ESP+SDA+FF there is a high variability in cost efficiency between plant sizes. This is due to low additional removal requirement (16,6 %), which results in an annual mercury removal of only 0,4 – 2,0 kg. Because

the absolute amount of removed mercury is so low, the variability in cost efficiency is emphasized.

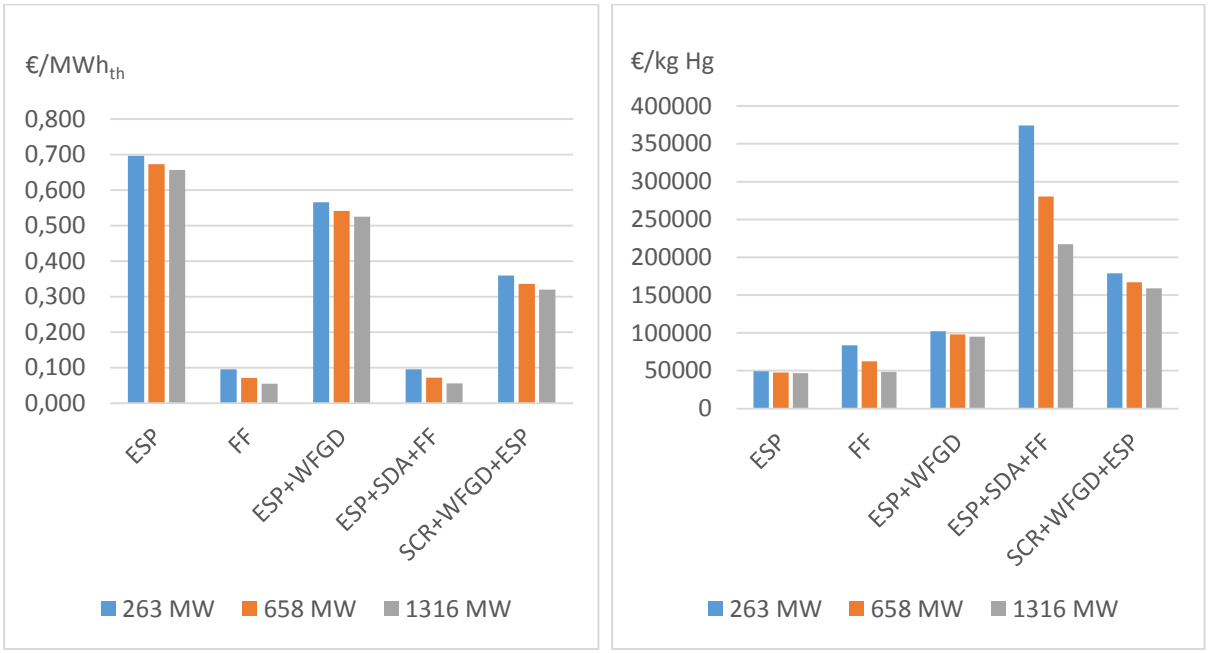

Figure 29: Annual costs for different flue gas cleaning configurations with the emission level 1 µg/Nm^3.

To demonstrate the structure of the costs, the Figures 30 and 31 show the annual costs (€/MWh$_{th}$) of plants with an ESP for emission levels 4 and 1 µg/Nm^3, respectively. It can be seen that variable operating costs are clearly dominating the total costs, mainly due to activated carbon costs. The share of activated carbon costs from the total costs is 86 – 95%, while the fixed operating costs are almost negligible when compared to others costs. In ESP plants, high additional reduction requirement leads to high consumption of activated carbon, which increases the operating costs significantly.

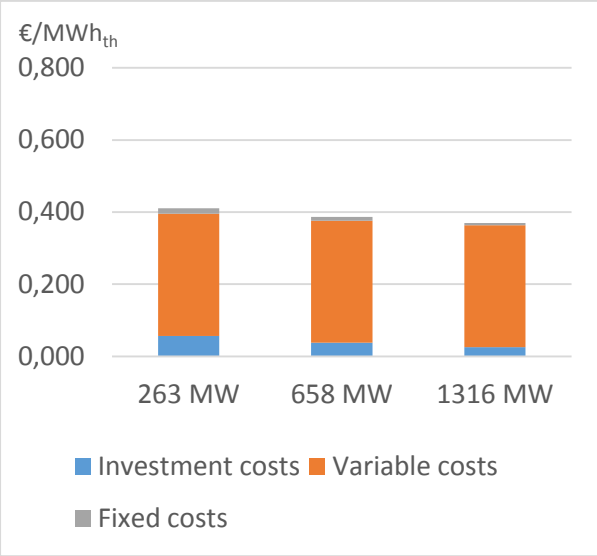

Figure 30: Structure of annual costs of activated carbon injection for ESP plants with the emission level 4 µg/Nm^3.

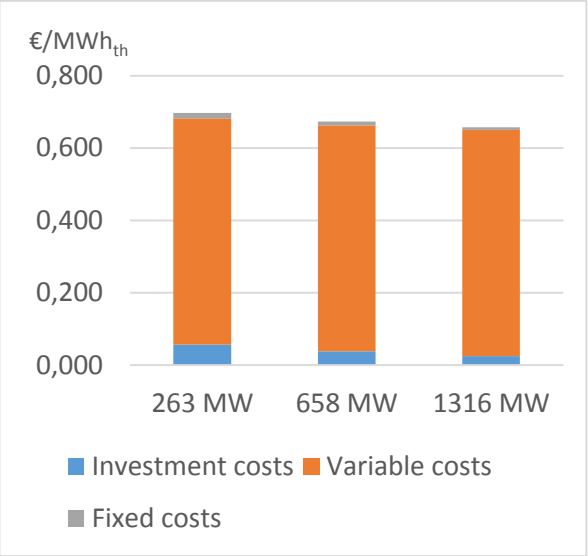

Figure 31: Structure of annual costs of activated carbon injection for ESP plants with the emission level 1 µg/Nm^3.

As a comparison, the Figure 32 shows the annual costs (€/MWh$_{th}$) of plants with a FF for emission level 1 µg/Nm^3. In this case, a lower additional removal rate of 47,0 % and better performance of FF in activated carbon injection systems lead to very low activated carbon consumption, which leads to low operating costs. Thus, the overall costs are much lower and the cost structure is different. The share of activated carbon form the total costs is only 22 – 39%. It shall be noted that because the scale of the y-axis is different to the Figures 30 and 31, the graphics are not directly comparable.

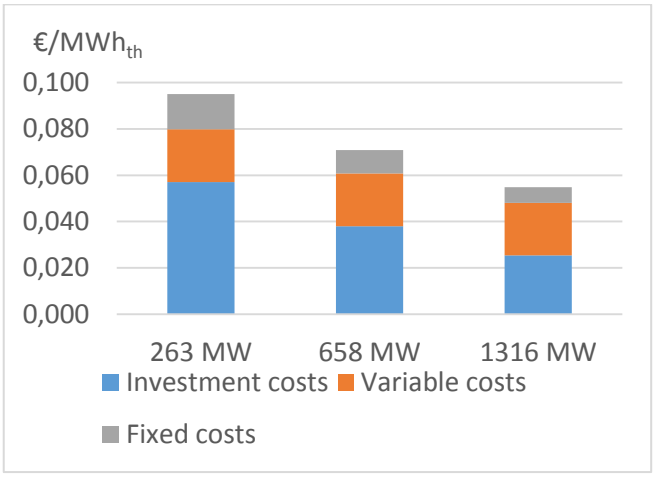

Figure 32: Structure of annual costs of activated carbon injection for FF plants with the emission level 4 µg/Nm^3.

9.4.3 Brominated activated carbon injection

Investment costs
The equipment for brominated activated carbon injection system is the same than for untreated activated carbon injection system. It consists of (Granite, Pennline et al. 2014)

- storage silo
- screw feeders (conveyors)
- metering system
- injection system (including metering, blowers, manifolds and lances).

Because the only difference between a brominated ACI system and untreated ACI system is the sorbent type, the investments costs are estimated to be the same. Some savings could be possibly made by a smaller storage silo, because the consumption rates of brominated activated carbon are usually much lower.

Operating costs
The operating costs comprise the same factors than in untreated activated carbon injection:

- variable costs:
  - brominated activated carbon
  - activated carbon disposal, if any
  - electricity
- fixed costs:
  - maintenance

The cost of brominated activated carbon is a little higher than the cost of untreated activated carbon, but the consumption rates are usually much lower. This is due to better mercury oxidation, which enhances the adsorption. The benefits of brominated activated carbon are best seen when low halogen fuels are used.

When compared to untreated activated carbon, the only difference in the operating costs is assumed to be the activated carbon itself. According to Sloss (2008), the prices of brominated activated carbons vary between 1,68 – 1,88 €/kg (1,89 - 2,11$/kg). This study will rely on the more recent information of the report provided by Reissner, Crèvecoeur et al. (2015), where a price of 1,78 €/kg was presented. Electricity consumption and maintenance cost is regarded as nearly the same than for untreated ACI. Additionally, because the consumption rate of brominated activated carbon is lower than the consumption rate of untreated activated carbon, the unburned content of the fly ash remains low. Because the brominated activated carbon is not expected to affect fly ash quality, fly ash disposal costs are not occurred.

The consumption values of operating costs are shown in the Table 33. The consumption of brominated activated carbon is calculated by the equation (23), and it expresses the injection rate that is needed for each flue gas cleaning configuration in order to achieve the required emission level. The equation (23) has different constants for plants with an ESP and plants with a FF, implicating that plants with a FF are more efficient to remove mercury. The relation

between activated carbon consumption and required removal rate is also shown in the Figure 25, with the exception that the minimum injection rate in this study is regarded as 5 mg/Nm³. All the studied plant configurations in this cost analysis have either an ESP or a FF, and the corresponding constants are applied for them. It is assumed that other air pollution control devices do not have influence on the activated carbon requirement.

Table 33: Consumption values for operating costs of brominated activated carbon injection.

	263 MW_{th}			658 MW_{th}			1316 MW_{th}		
Emission level (µg/Nm³)	9	4	1	9	4	1	9	4	1
Brominated activated carbon consumption (kg/h)									
- ESP	7,0	22,2	44,5	17,4	55,5	111,2	34,9	111,0	222,4
- FF	-	-	1,8	-	-	4,6	-	-	9,2
- ESP+WFGD	-	6,9	33,2	-	17,3	83,0	-	34,7	166,1
- ESP+SDA+FF	-	-	1,8	-	-	4,6	-	-	9,2
- SCR+ESP+WFGD	-	-	19,0	-	-	47,5	-	-	95,0
- SCR+ESP+SDA+FF	-	-	-	-	-	-	-	-	-
Electricity (kW)	6,5			16,2			32,4		
Maintenance (%/investment costs)	3			3			3		

Total costs

The summary of total costs of brominated activated carbon injection for plant sizes 263 – 1316 MW_{th} is shown in the Table 34. The Table shows the cost range for different existing air pollution control configurations and for different target emission levels. The cases that do not require additional mercury removal are excluded from the Table 34, and the costs for them is zero. As with the untreated ACI, the annual costs and the increase in electricity price is higher when higher additional removal rate is required. Also in this case, the cost efficiency (€/kg Hg removed) may be poor if the required additional removal rate is low. The reason for this is the low amount of removed mercury.

Table 34: Summary of the total costs of brominated activated carbon injection for different emission levels.

	Required additional removal (%)	Annual costs (€/MWh$_{th}$)	Increase in electricity price (cent/kWh$_e$)	Cost efficiency (€/kg Hg)
9µg/Nm^3				
- ESP	25	0,081 – 0,121	0,021 – 0,032	21200 – 31 600
4µg/Nm^3				
- ESP	66,7	0,184 – 0,224	0,048 – 0,059	18 000 – 21 900
- ESP+WFGD	24,7	0,081 – 0,121	0,021 – 0,032	48 200 – 72 100
1µg/Nm^3				
- ESP	91,7	0,335 – 0,375	0,088 – 0,099	23 800 – 26 600
- FF	47	0,047 – 0,087	0,012 – 0,023	41 000 – 76 400
- ESP+WFGD	81,2	0,259 – 0,299	0,068 – 0,079	46 800 – 54 100
- ESP+SDA+FF	16,6	0,047 – 0,087	0,012 – 0,023	181 900 – 338 800
- SCR+ESP+WFGD	61,1	0,163 – 0,203	0,043 – 0,053	80 900 – 100 800

The annual costs and cost efficiency of brominated activated carbon injection for different air pollution configurations and plant sizes are illustrated in the Figure 33. The applied emission level is 1 µg/Nm^3. The graphics have similar structure to those of untreated activated carbon injection. However, the differences between plant capacities seem to be little higher.

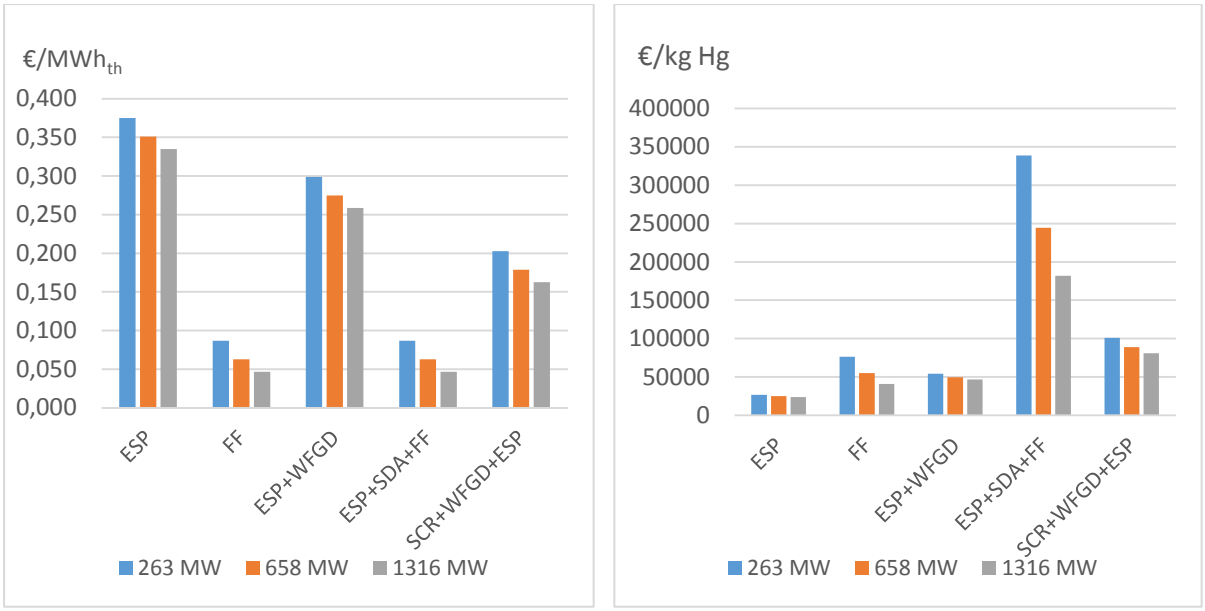

Figure 33: Annual costs of brominated activated carbon injection for different flue gas cleaning configurations with the emission level 1 µg/Nm^3.

To further demonstrate the cost structure of brominated ACI, the Figures 34 and 35 show the annual costs (€/MWh$_{th}$) divided in investment costs, variable operating costs and fixed operating costs. The air pollution control configuration is ESP only, and the applied emission levels are 4 and 1 µg/Nm^3. As with the untreated ACI, the variable operating costs comprise by far the biggest share of the total costs, mainly due to activated carbon costs. The share of the activated carbon from the total costs is 87 – 90%.

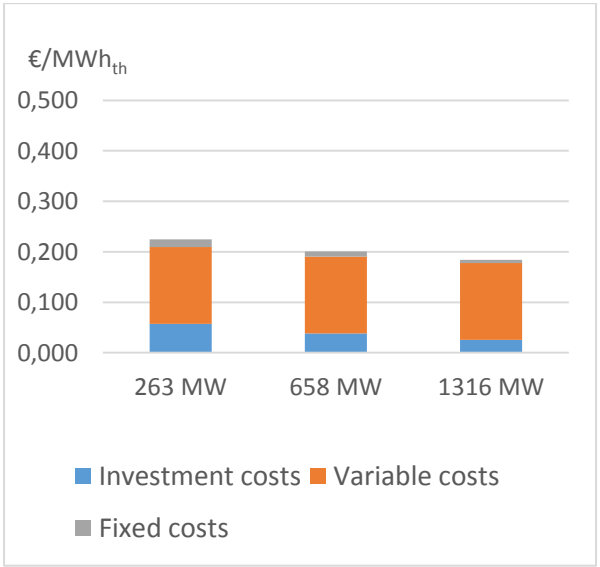

Figure 34: Structure of annual costs of brominated activated carbon injection for ESP plants with the emission level 4 µg/Nm^3.

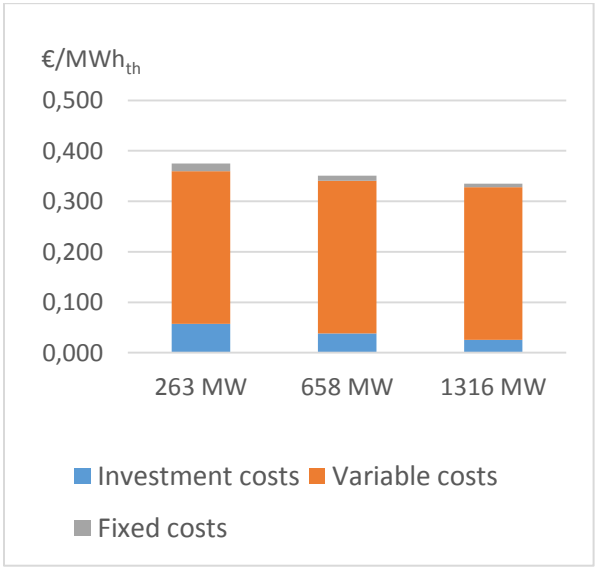

Figure 35: Structure of annual costs of brominated activated carbon injection for ESP plants with the emission level 1 µg/Nm^3.

9.4.4 Bromine injection

Investment costs

The investment costs of a halogen injection system consist of the purchasing and installation of process equipment. The process equipment consist of bulk storage tank, metering, pumps, piping, and injection assemblies. The equipment cost is less expensive, if the injection point is installed upstream of the power plant process. (Granite, Pennline et al. 2014)

Some data about the costs of $CaBr_2$ injection system is provided by US National Energy Technology Laboratory (NETL). NETL has conducted field test at a 500 MW_e Monticello plant in the USA. The system consisted of a storage tank, four transfer pumps, piping, flowmeters and controllers. The costs for preparation, installation, piping and electrical connection of the system were approximately 694 000€ (780 000 $), which makes 1,39€/$kW_e$. (Jones, Feeley III 2008)

More recent information is given by the URS report, according to which the capital costs are between 2,23 – 3,12 €/kW_e (2,5 – 3,5 $/$kW_e$). The average capital cost of installation is estimated to be 2,67 €/kW_e (3 $/kW). (Lipinski, Leonard et al. 2011)

In this study, the average price of 2,67 €/kW_e is used for the middle-sized plant (250 MW_e) and this price is scaled for the plants 100 MW_e and 500 MW_e by using the equation (21). Because value for the capacity factor e is not known, the general value of 0,6 is used. This yields capital costs of 4,62 €/kW_e for plants 100 MW_e and 1,76 €/kW_e for plants 500 MW_e. The results are showed in the Figure 36.

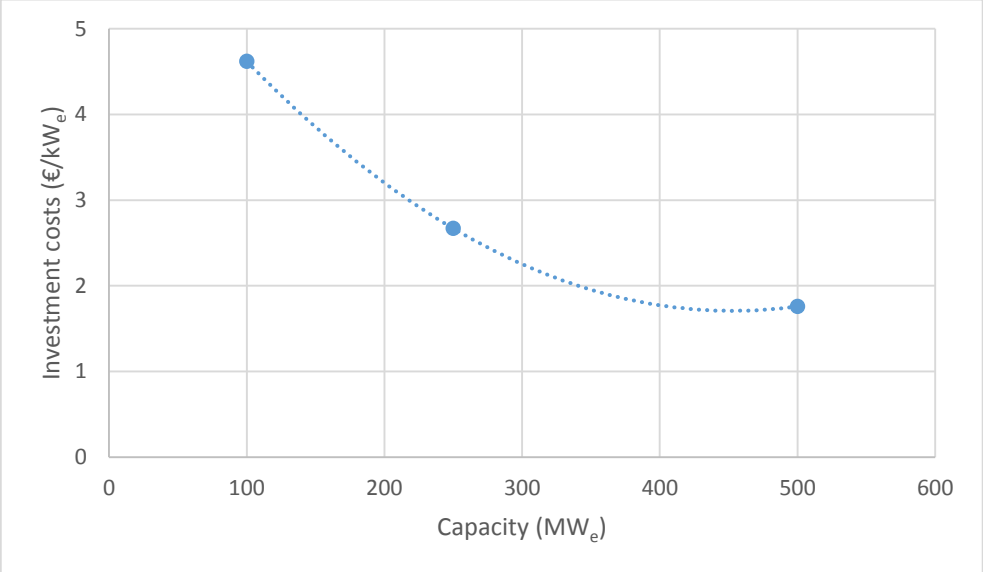

Figure 36: Investment costs of halogen injection system for different plant sizes.

Operating costs
The operating costs of a bromine injection system are following:

- variable costs:
    - chemical additives
    - electricity
- fixed costs:
    - maintenance

The greatest share of operating costs consist of halogen additive itself. There are many types of halogen based products on the market. Usually, bromine based products are used, such as calcium bromide ($CaBr_2$), sodium bromide (NaBr) and hydrobromic acid (HBr). Most used product is calcium bromide as a 52 wt% solution. Sodium bromide is also used, but the molar ratio of sodium and bromine is higher (1:1) than the molar ratio of calcium bromide (1:2). This means that sodium bromide consumption is higher than the consumption of calcium bromide. Moreover, hydrobromic acid is not so popular, mainly due to suspicions of its acidic properties. In this study 52 wt% calcium bromide solution is used as the halogen additive. (Lehmkuhler 2016)

The total operating costs depends much on the price and consumption rate of calcium bromide. According to Lehmkuhler (2016), the costs for 52 wt% calcium bromide solution vary between 1,5 – 1,75 €/kg. The halogen price depends also notably on the distance between the supplier and power plant, also transport cost. In the URS report, a price of 1,96 €/kg (1 $/lb) is given for 52 wt% calcium bromide. This price is also used in this study, and it assumed to include the transportation costs. Because 1kg of bromine corresponds 2,50 kg of 52 wt% $CaBr_2$ –solution, the price for pure bromine is 4,90 €/kg. Nevertheless, the market price of calcium bromide varies a lot, and it is hard to estimate if there will be an increase in the future. For example, the price was more than doubled between the years 2004 – 2011. (Lipinski, Leonard et al. 2011)

Most importantly, the required amount of additional halogen depends on the natural halogen content of the fuel. If the fuel is rich in halogen, the initial halogen content is usually high enough to provide enough mercury oxidation, and less additional halogen is needed. Also, if SCR is present in the configuration, sufficient oxidation is occurred already with low halogen consumption. According to Lehmkuhler (2016), in general, 90% removal rates are achieved already with a bromine injection rate of 20 – 25 ppm in fuel, when SCR is present. Without SCR, bromine injection rates of 100 – 150 ppm are required to achieve the same removal rate.

Because there is no similar algorithm for estimating the halogen consumption as there was for estimating the activated carbon consumption rates in chapters 9.4.2 and 9.4.3, this study will rely on field test data found in the literature. However, there is not much field test data is available on the relation between halogen injection rate and mercury removal. Most of the existing data is from tests made in US power plants, and usually they are based on tests at power plants that burn either lignite or subbituminous coal. Some test data is presented in the Figure 37 and the Table 35. The injected chemical is calcium bromide and the values are presented as ppm of pure bromine in the fuel. As can be seen, in the plants that are equipped with an SCR,

90% removal rates can be achieved by injection rate of 25 ppm. If SCR is not installed, increased injection rates are required.

To approximate the injection rates in this analysis, an estimate curve was generated on the basis of the field test data. The estimate curve is shown in the Figure 37, and it shows the approximated correlation between mercury removal and bromine injection rate. The estimate curve applies to plants without an SCR. For plants with SCR, it is assumed that 25 ppm injection rate is sufficient to meet any removal rate. Thus, it also assumed that for plants without SCR, 25 ppm is the minimum injection rate that is used for any mercury removal. For example, as can be seen from the Figure 37, bromine injection rate of 250 ppm is required for removal rate of 90 %, and injection rate of 50ppm is needed for 60 % removal.

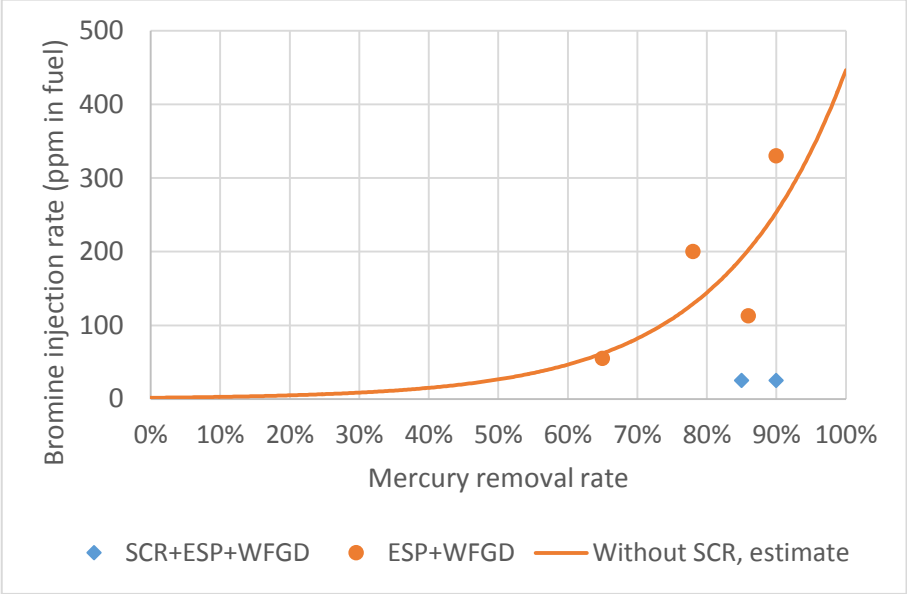

Figure 37: Bromine injection rate as a function of targeted mercury removal rate.

Table 35: Field test data related to the Figure 37.

	Plant	Air pollution control	Coal	Source
1	Monticello, USA	ESP + WFGD	Subbituminous/Lignite	(Lipinski, Leonard et al. 2011)
2	Monticello, USA	ESP + WFGD	Subbituminous/Lignite	(Benson, Holmes et al. 2007)
3	Monticello, USA	ESP + WFGD	Subbituminous/Lignite	(Feeley III, Jones et al. 2009)
4	Monticello, USA	ESP + WFGD	Subbituminous/Lignite	(Feeley III, Jones et al. 2009)
5	Plant Miller, USA	SCR + ESP + WFGD	Subbituminous/Lignite	(Feeley, Jones 2008)
6	Pleasant Prairie, USA	SCR + ESP + WFGD	Bituminous	(Vosteen, Hartmann et al. 2012)

As explained in the earlier chapters, halogen injection aims to increase the share of oxidized mercury, so it can be removed in FGD system. Thus, it assumed that plants that that have only ESP or FF, halogen injection is not applicable technology for mercury removal. The bromine consumptions for each configuration and for each emission level are shown in the Table 36.

Electricity is required mainly for the injection pump. Also, the metering and control system requires some electricity. Usually, the electricity consumption is so low that it can be excluded from the cost estimations. (Lehmkuhler 2016). As an example, with bromine injection rate of 200 ppm, the volume flow of 52 wt% calcium bromide solution for the largest power plant (1316 MW_{th}) would be 0,064 m^3/h, also 0,018 dm^3/s. The pump power consumption can be calculated by the equation (25). (Wiksten 2009)

$$P_s = \frac{q\rho g h}{\eta} \tag{25}$$

Where,

P_s is the pump power
q is the volume flow of the fluid
ρ is the density of the fluid
g is the gravity (9,81 m/s^2)
h is the differential head
η is the pump efficiency

The density of 52 wt% $CaBr_2$ solution is approximately 1700kg/m^3 (GLS 2009). With initial parameters of q = 0,018 * 10^{-3} m^3/s, ρ = 1700 kg/m^3, h = 40m and η = 70%, the pump power consumption would be 0,017 kW, so the electricity consumption can be neglected.

Table 36: Consumption values for operating costs of bromine injection.

	263 MW_{th}			658 MW_{th}			1316 MW_{th}		
Emission level (µg/Nm³)	9	4	1	9	4	1	9	4	1
Calcium bromide (52wt-%) consumption (kg/h)									
- ESP	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
- FF	-	-	N/A	-	-	N/A	-	-	N/A
- ESP+WFGD	-	2,5	15,0	-	6,2	37,5	-	12,5	75,0
- ESP+SDA+FF	-	-	2,5	-	-	6,2	-	-	12,5
- SCR+ESP+WFGD	-	-	2,5	-	-	6,2	-	-	12,5
- SCR+ESP+SDA+FF	-	-	-	-	-	-	-	-	-
Electricity (kW)	-			-			-		
Maintenance (%/investment costs)	3			3			3		

Total costs

Summary of the costs of bromine injection for plant sizes 263 – 1316 MW_{th} is shown in the Table 37. The costs are shown for plants that have either WFGD or SDA. For those plants the bromine injection is an applicable technology, because oxidized mercury can be removed in the desulphurization systems. The cases that do not require additional mercury removal are excluded from the Table 37, and the costs for them is zero.

Table 37: Summary of the total costs of bromine addition for different emission levels.

	Required additional removal (%)	Annual costs (€/MWh_{th})	Increase in electricity price (cent/kWh_e)	Cost efficiency (€/kg Hg)
9µg/Nm^3				
- ESP	25	N/A	N/A	N/A
4µg/Nm^3				
- ESP	66,7	N/A	N/A	N/A
- ESP+WFGD	24,7	0,034 – 0,060	0,009 – 0,016	20 500 – 35 800
1µg/Nm^3				
- ESP	91,7	N/A	N/A	N/A
- FF	47	N/A	N/A	N/A
- ESP+WFGD	81,2	0,128 – 0,153	0,034 – 0,040	23 100 – 27 800
- ESP+SDA+FF	16,6	0,034 – 0,060	0,009 – 0,016	134 800 – 235 700
- SCR+ESP+WFGD	61,1	0,034 – 0,060	0,009 – 0,016	17 100 – 30 000

The annual costs and cost efficiency of bromine injection for different air pollution configurations and plant sizes are illustrated in the Figure 38. The applied emission level is 1 µg/Nm^3. In general, the costs are higher for plants that have higher additional removal requirement and that do not apply SCR, because more bromine is required. Thus, the highest annual costs are occurred to ESP+WFGD configuration. The annual costs for ESP+SDA+FF and SCR+ESP+WFGD are more than 50% lower than the costs for ESP+WFGD, because the bromine consumption is low. It can be also noted that annual costs for ESP+SDA+FF and SCR+ESP+WFGD are the same. This is because the minimum bromine consumption was determined as 25ppm, and it was also estimated that for plants that have SCR, the bromine consumption is 25ppm regardless of the removal rate. This leads to same operating costs for these two configurations.

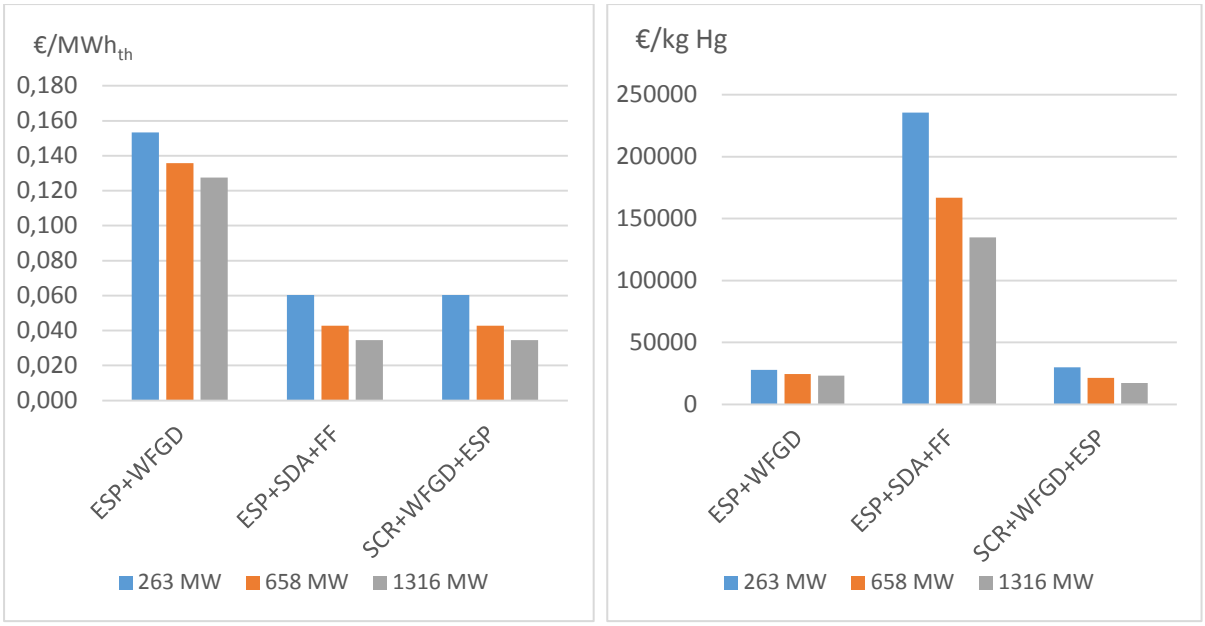

Figure 38: Annual costs of bromine injection for different flue gas cleaning configurations with the emission level 1 µg/Nm^3.

The cost structure of bromine injection for emission levels of 4 and 1 µg/Nm^3 is shown in the Figures 39 and 40, respectively. The applied initial air pollution control is ESP+WFGD. Also with bromine injection, the operating costs are dominating, but not that dominating as with activated carbon injection. With the emission level of 4 µg/Nm^3, $CaBr_2$ –solution comprises about 45 - 64% of the costs. If the emission level is 1 µg/Nm^3, more bromine is needed and $CaBr_2$ –solution comprises 73 – 88% of the total costs.

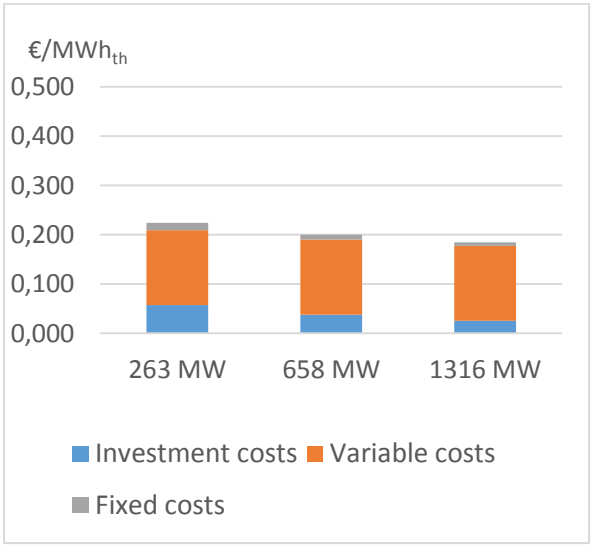

Figure 39: Structure of annual costs of bromine injection for ESP+WFGD plants with the emission level 4 µg/ Nm^3.

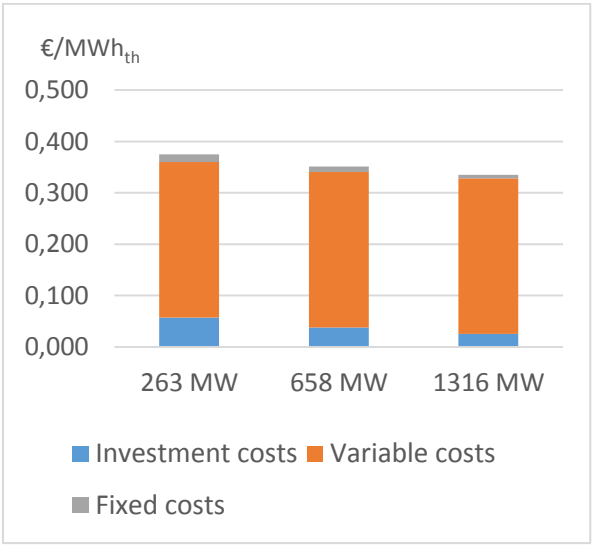

Figure 40: Structure of annual costs of bromine injection for ESP+WFGD plants with the emission level 1 µg/Nm^3.

10 Results and discussion

In the chapter 9, an analysis was made to determine the costs of the most common mercury-specific removal technologies: untreated activated carbon injection, brominated activated carbon injection and bromine injection. Cost estimates were made for existing plants that have different flue gas cleaning configurations and different capacities. Furthermore, calculations were made for three different mercury emission levels, which were 9, 4 and 1 µg/Nm^3. One of the main goals of this study was to determine the costs of different mercury removal technologies that are occurred when these emission levels are achieved. In this chapter the costs of the technologies are analysed and compared.

10.1 Comparison of the costs

The summary of annual costs (€/MWh$_{th}$) and cost efficiency (€/kg Hg removed) of untreated activated carbon injection, brominated activated carbon injection and bromine injection for different plant configurations are shown in the Figures 41, 42 and 43, respectively. Costs are shown for the strictest emission level of 1 µg/Nm^3. Costs of bromine injection are represented only for the ESP+WFGD, ESP+SDA+FF and SCR+ESP+WFGD configurations, because it is feasible technology only in plants that have a desulphurization system. The sixth configuration that was included in the study, SCR+ESP+SDA+FF, is not shown in the Figures, because no additional removal was required. Costs of all three studied removal technologies for all emission levels and plant configurations are summarized in the Table 38. The Table 38 shows also the cases where no additional mercury removal is required and costs are regarded as zero.

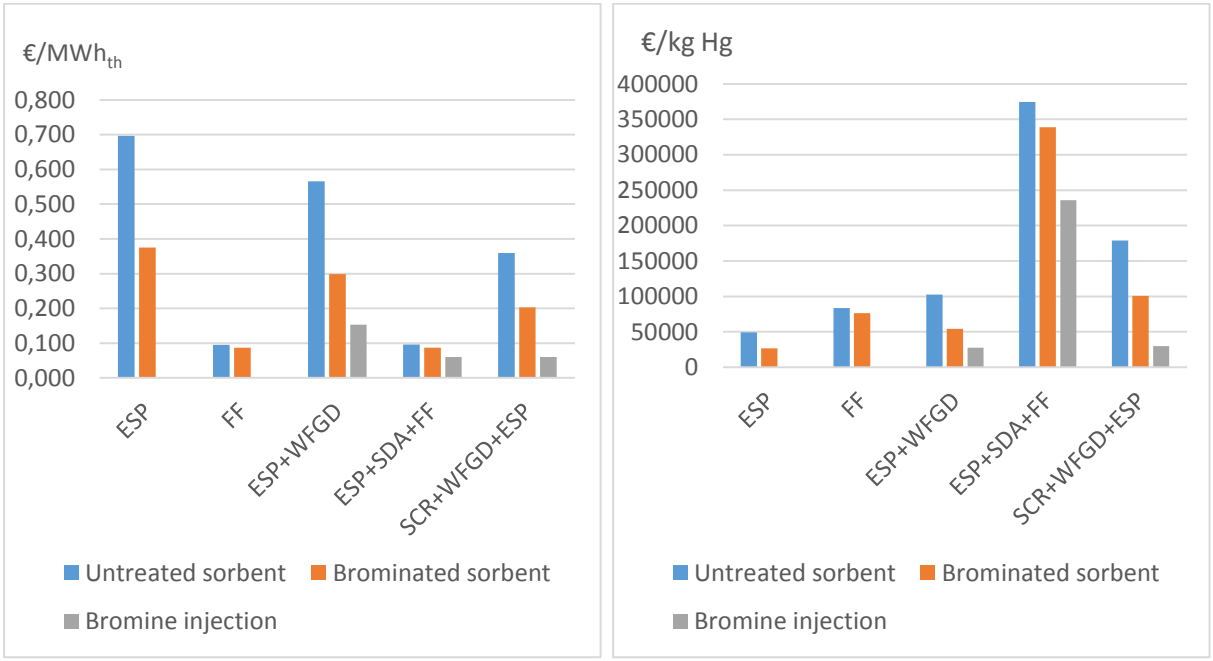

Figure 41: Summary of the costs for 263 MW$_{th}$ plants with the emission level 1 µg/ Nm^3.

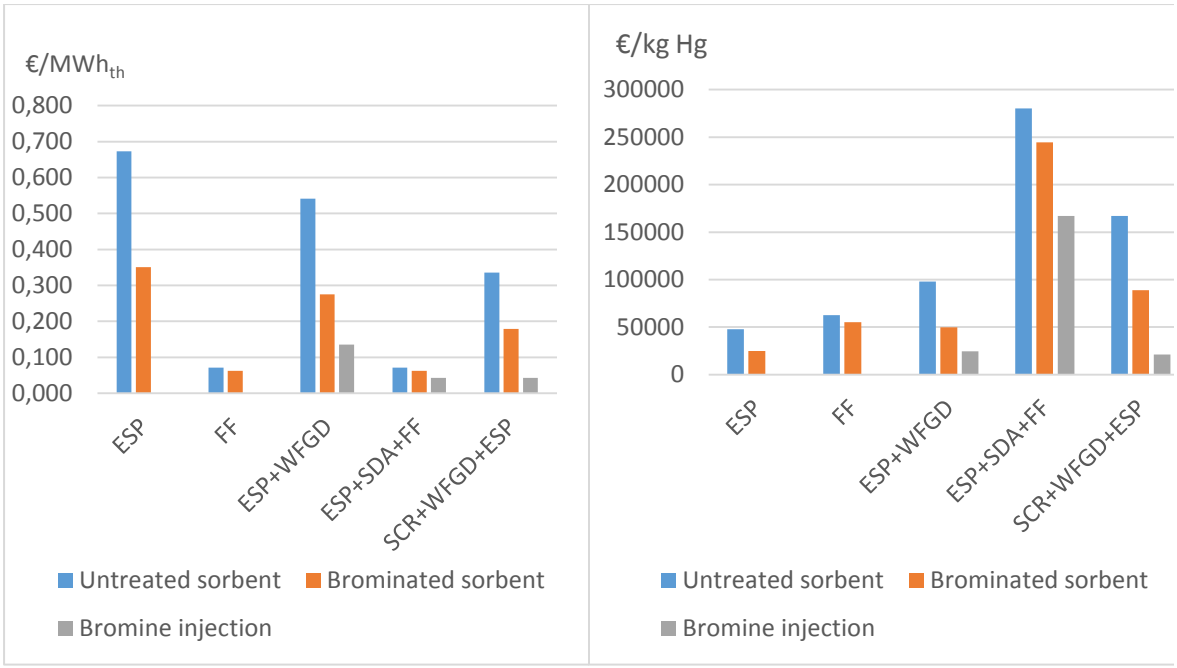

Figure 42: Summary of the costs for 658 MW_{th} plants with the emission level 1 µg/ Nm^3.

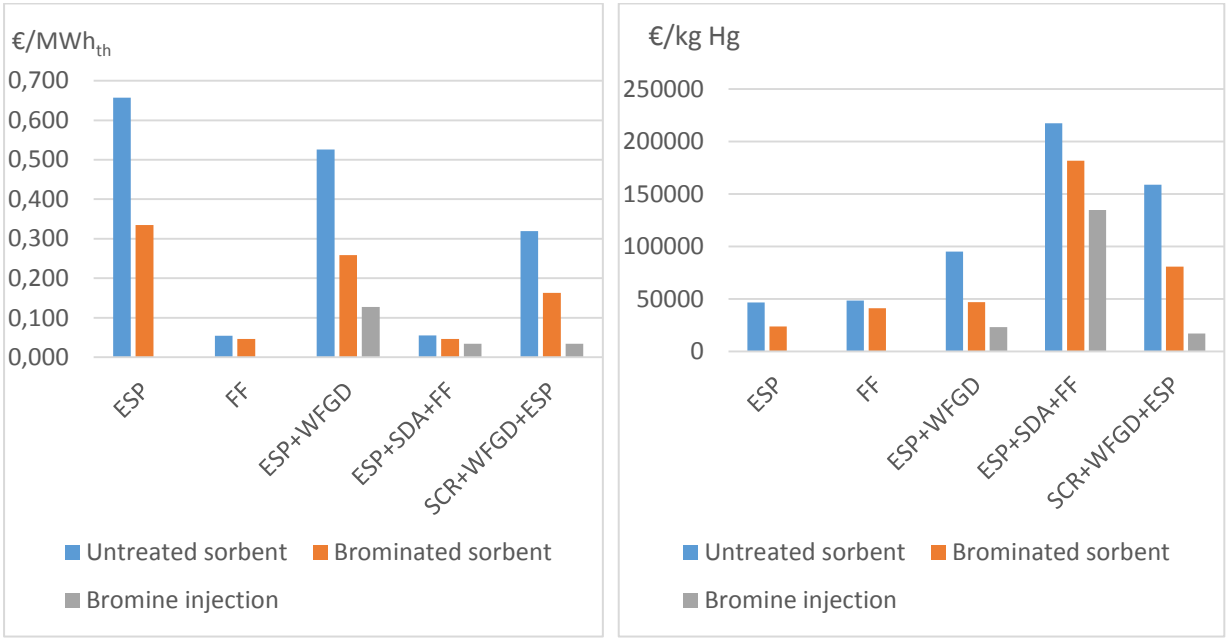

Figure 43: Summary of the costs for 1316 MW_{th} plants with the emission level 1 µg/ Nm^3.

Table 38: Summary of the mercury removal costs of untreated activated carbon injection, brominated activated carbon injection and bromine injection for different emission levels.

		Annual costs (€/MWh$_{th}$)			Cost efficiency (€/kg Hg)		
Existing APCD	Additional removal (%)	ACI	Brominated ACI	Bromine addition	ACI	Brominated ACI	Bromine addition
9µg/ Nm^3							
ESP	25	0,113 – 0,153	0,081 – 0,121	N/A	29 400 – 39 800	21 200 – 31 600	N/A
FF	-	-	-	-	-	-	-
ESP+WFGD	-	-	-	-	-	-	-
ESP+SDA+ FF	-	-	-	-	-	-	-
SCR+ESP+ WFGD	-	-	-	-	-	-	-
SCR+ESP+ SDA+FF	-	-	-	-	-	-	-
4µg/Nm^3							
ESP	66,7	0,370 – 0,410	0,184 – 0,224	N/A	36 100 – 40 100	18 000 – 21 900	N/A
FF	-	-	-	-	-	-	-
ESP+WFGD	24,7	0,112 – 0,152	0,081 – 0,121	0,034 – 0,060	66 500 – 90 400	48 200 – 72 100	20 500 – 35 800
ESP+SDA+ FF	-	-	-	-	-	-	-
SCR+ESP+ WFGD	-	-	-	-	-	-	-
SCR+ESP+ SDA+FF	-	-	-	-	-	-	-
1µg/ Nm^3							
ESP	91,7	0,657 – 0,697	0,335 – 0,375	N/A	46 600 – 49 500	23 800 – 26 600	N/A
FF	47	0,055 – 0,095	0,047 – 0,087	N/A	48 400 – 83 800	41 000 – 76 400	N/A
ESP+WFGD	81,2	0,526 – 0,566	0,259 – 0,299	0,128 – 0,153	95 200 – 102 400	46 800 – 54 100	23 100 – 27 800
ESP+SDA+ FF	16,6	0,056 – 0,096	0,047 – 0,087	0,034 – 0,060	217 500 – 374 400	181 900 – 338 800	134 800 – 235 700
SCR+ESP+ WFGD	61,1	0,320– 0,360	0,163 – 0,203	0,034 – 0,060	159 000 – 178 900	80 900 – 100 800	17 100 – 30 000
SCR+ESP+ SDA+FF	-	-	-	-	-	-	-

It seems that regardless of the plant capacity or required removal rate, the costs of the technologies have the following rank order from the cheapest to the most expensive one: bromine injection, brominated activated carbon injection and untreated activated carbon injection. The rank order is the same irrespective of whether the focus is on the annual costs (€/MWh_{th}) or on the cost efficiency (€/kg Hg). The low costs of bromine injection are probably explained by lower investment cost and lower chemical consumption. In bromine injection, the equipment is very simple and does not require much capital.

The second most cost-effective alternative is brominated activated carbon injection. The equipment in brominated ACI and untreated ACI is similar, so the investment costs are the same. Thus, the explanation for the difference in the annual costs can be drawn to operating costs. Although brominated activated carbon is more expensive than untreated activated carbon, it is likely that the lower consumption rates of brominated activated carbon can overcome the cost difference.

The variation of the annual costs (€/MWh_{th}) within different plant sizes is high in some cases and low in another ones. In general, the variation depends on the share of operating costs in total costs. High share of operating costs result in low variation of annual costs within plant sizes, and vice versa. The operating costs of the studies technologies consist mainly of the sorbent/halogen cost, which in turn depends on the consumption. The consumption is higher when higher removal rate is required. Thus, the variation of the costs is high in plants that have low additional reduction requirement, and low operating costs.

This can be seen in ESP plants, where 91,7% reduction is required at the emission level 1 μg/Nm^3. With untreated ACI, the annual cost variation is only 0,657 – 0,697 €/MWh_{th} within plant sizes. The annual costs of 263 MW_{th} plants are almost the same than the annual costs of 1316 MW_{th} plants. In this case, the share of operating costs of the total costs is 92 – 96%.

On the other hand, plants with ESP+SDA+ FF have only 16,6% reduction requirement at the emission level 1 μg/Nm^3. With untreated ACI the relative annual cost variation is much higher: 0,056 – 0,096 €/MWh_{th}. Annual costs of 263 MW_{th} plants are almost twice as high as the annual costs of 1316 MW_{th} plants. In this case, the share of operating costs of total costs is only 40 – 54%.

In general, the capital costs of the studied technologies are low and most of the costs are caused by the consumed sorbent/halogen. In most cases where significant reduction rates are achieved, the operating costs comprise up to 90% of the total costs. Thus, the economics of scale does not appear so strongly in the studied technologies.

The same effect can be seen in the cost efficiency (€/kg Hg) when different plant sizes are compared. This is because mercury is removed in very low amounts, when the required additional reduction is low. As the plant size doubles, the absolute amount of mercury doubles as well, although the costs are not increased that much. This causes the variation in cost efficiency.

According to this analysis, the costs of mercury removal are quite low. The range in annual cost of the studied cases is 0,034 – 0,697 €/MWh_{th}. The increase in electricity price would be 0,009 - 0,183 cents/kWh. The costs depend on the existing flue gas cleaning configuration (also baseline removal rate), mercury emission limit, plant size and chosen technology. Lower baseline mercury removal, stricter mercury emission limit and smaller plant size lead to higher costs.

The highest annual cost of 0,657 – 0,697 €/MW would occur for plants that have only ESP and that apply untreated ACI with a target emission level of 1 μg/Nm^3. The high additional mercury removal rate of 91,7% increases the costs. The increase in electricity price is 0,173 – 0,183 cents/kWh. The lowest cost of 0,034 – 0,060 €/MWh_{th} would occur for plants that has either configuration ESP+SDA+FF or SCR+ESP+WFGD and that apply bromine injection with a target emission level of 1 μg/Nm^3. The increase in electricity price would be 0,009 – 0,016 cents/kWh. Those cases, the additional removal requirement is 16,6% or 61,1%.

The worst cost efficiency of 217 500 – 374 400 €/kg Hg is occurred for a plant that has a configuration ESP+SDA+FF and that apply untreated ACI with a target emission level of 1 μg/Nm^3. This is explained by a low additional removal requirement of 16,6%. Removal requirement is so low that not much mercury is removed during the operation of the plant and regardless of the very low operating costs, the investment costs will increase the price of a removed mercury kilogram.

The best cost efficiency of 18 000 –21 900 €/kg Hg is occurred in plants that have only ESP and that apply brominated ACI. Interestingly, this is achieved with a target emission level of 4 μg/Nm^3. When emission level is increased to 9 μg/Nm^3, the cost efficiency drops to 23 800 – 26 600 €/kg. The high cost efficiency of ESP plants is naturally explained by high additional removal. Regardless of the high operating costs, the amount of removed mercury is so large that cost efficiency remains good. When the emission level is tightened from 4 to 1 μg/Nm^3, the increase in activated carbon consumption is so high that cost-efficiency decreases.

A cost comparison between untreated ACI and brominated ACI are shown in the Figures 44 (€/MWh_{th}) and 45 (€/kg Hg removed). The costs are represented for different capacities for plants that have only an ESP. As can be seen from the Figure 44, the annual costs increase as the required removal rate increases. In general, the costs of brominated activated carbon are lower. However, at low removal rates, there is not much cost difference between untreated and brominated activated carbon, and the costs are almost equal. As the removal rate increases, the cost gap between the sorbents increases as well. The explanation for this is probably the performance difference between the untreated and brominated activated carbon. The consumption of untreated activated carbon increases more rapidly than the consumption of brominated one, as the removal rate is increased. This leads to higher operating costs with untreated ACI, which is emphasized at higher removal rates.

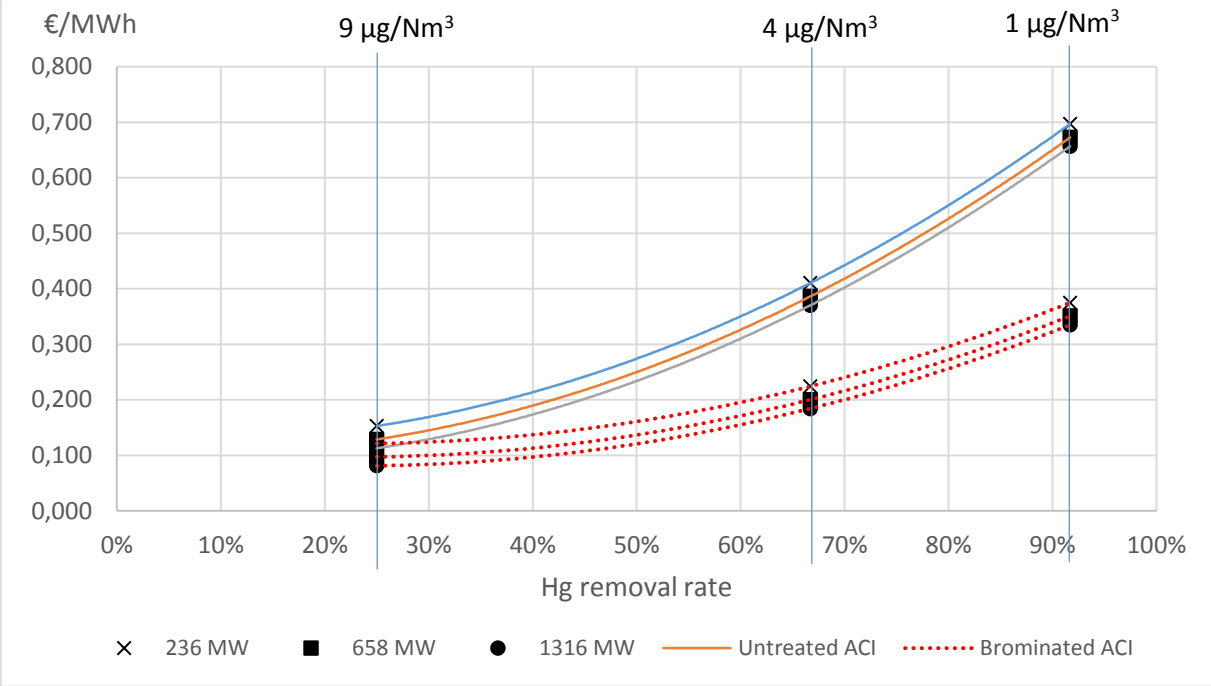

Figure 44: Comparison of annual costs (€/MWh$_{th}$) between untreated ACI and brominated ACI for ESP plants.

The Figure 45 verifies the lower costs of brominated ACI. An interesting observation is the improvement in the cost efficiency of brominated ACI when the removal rate is increased from 25% to 70%. After that the cost per removed kilogram start to increase again. A slight improvement can also be seen for untreated ACI at the 263 MW$_{th}$ plant size, but the phenomenon is not so strong. Again, the explanation for this has to be the better performance of brominated ACI, which is emphasized at higher removal rates. However, after 70%, the ratio between spent activated carbon and removed mercury increased so much that the cost efficiency starts to decrease. When comparing the plant sizes, there seems to be more difference in cost efficiency at lower removal rates. As the removal rates are increased, the cost efficiency of different plant sizes gets closer to each other.

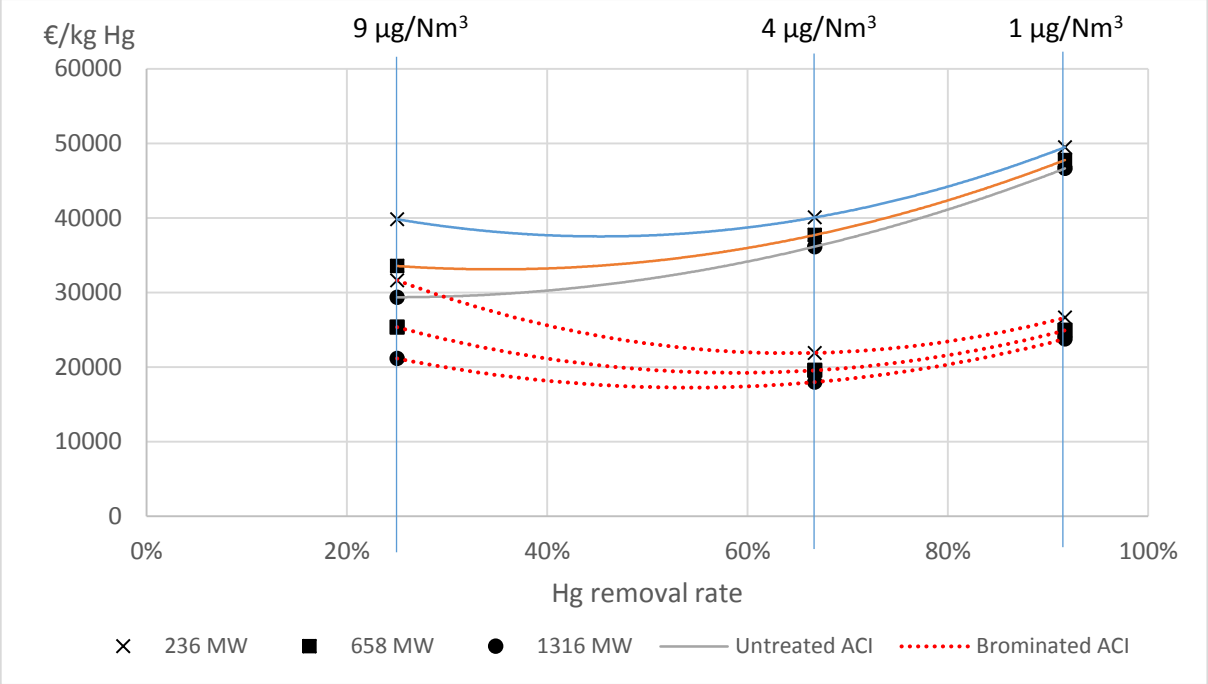

Figure 45: Comparison of cost efficiency (€/kg Hg removed) between untreated ACI and brominated ACI for ESP plants.

The results of this study were also compared to other studies that have examined the costs of mercury removal. In an economic study made by Sloss (2008), information about mercury removal costs has been gathered together. Some of cost information about activated carbon injection at different removal rates are shown the Table 39. The data represented in the Table 39 originate from cost analyses that were conducted for full-scale plants in the USA. All of the reference plants of the Table 39 burn bituminous coal and are equipped at least with an ESP.

Table 39: Mercury removal costs at different removal rates for bituminous coal-fired power plants. (Sloss 2008)

Plant	Sorbent type	Baseline Hg removal	Hg removal rate 50 %		Hg removal rate 70 %		Hg removal rate 80-90%	
		(%)	Cent/ kWh	€/kg Hg	Cent/ kWh	€/kg Hg	Cent/ kWh	€/kg Hg
Yeates Unit 1	(Super HOK) Untreated sorbent	50	0,09	108082	0,15	136081	N/A	N/A
Monroe Unit 4	(DARCO® Hg) Brominated sorbent	25	0,03	33678	0,07	46992	0,11	66180
Lee Unit 1	(B-PACTM) Brominated sorbent	20	0,10	139801	0,17	170738	0,26	201674
Portland Unit 1	(Mer-Clean™ 8-21) (Treated sorbent)	30	0,04	26237	0,06	29174	0,17	63243

The costs shown in the Table 39 are of the same order of magnitude than the costs obtained in this study. The increase in electricity price varies between 0,03 – 0,26 cent/kWh and the cost efficiency between 26 200 – 201 800 €/kg depending on the baseline removal rate and additional removal rate. In our study, the values were 0,009 - 0,183 cent/kWh and 18 000 – 374 400 €/kg Hg.

For example, in our study, the baseline mercury removal rate of ESP was determined as 30%. An additional removal of 66,7% by brominated ACI resulted in costs of 0,048 – 0,059 cent/kWh or 18 000 – 21 900 €/kg Hg. This can be compared to the plant Monroe 4 that has a baseline mercury removal rate of 25 %. At removal rate of 70%, the costs of brominated injection are reported to be 0,07 cent/kWh or 46 992 €/kg Hg. The increase in electricity price is similar, but the cost for removed mercury is twice as high in the Monroe 4 plant.

At removal rate of 80-90%, the corresponding values are 0,11 cent/kWh and 66180 €/kg Hg in the Monroe Unit 4. In our study, removal rate of 91,4% resulted in costs of 0,088 – 0,099 cent/kWh and 23 800 – 26 600 €/kg Hg. Again, the increase in electricity price is similar, but the price difference in cost efficiency (€/kg Hg) is bigger. This can be explained by several factors, such as different mercury content of coal.

Similar costs are reported in Portland Unit 1, which also uses a treated sorbent and that has the same baseline removal rate of 30% than our ESP case. However, in the Lee Unit 1 the costs are much higher, although brominated sorbent is used also there. This reminds that mercury removal costs are very site-specific and depend on various factors.

10.2 Sensitivity analysis

Because there are so many factors affecting the mercury removal cost, a sensitivity analysis was made to investigate the impact of certain parameters. As a reference case, the plant configuration ESP only and the target emission level 1 µg/Nm^3 is chosen. The technology chosen is brominated ACI.

As explained earlier, activated carbon cost and consumption has a huge impact on total costs, because variable operating costs may comprise up to 90 of the total costs. It was also mentioned before that the activated carbon cost may vary a lot on the market. This is why the total costs are sensitive to activated carbon cost variation. The Figure 46 illustrates the influence of activated carbon price on the total annual costs of ACI system. The brominated activated carbon price that was initially used in the analysis was 1,78 €/kg, and the total annual costs were 0,335 – 0,375 €/MWh$_{th}$ depending on the plant size. It can be seen that the linear dependency between activated carbon cost and total annual costs. The impact of activated carbon cost is high, as an increase or decrease of 50% in activated carbon price leads to increase or decrease of 43-45 % in total annual costs.

Furthermore, the number of operating hours of a power plant may vary. The impact of operating hours is illustrated in the Figure 47. The initial operating hours was 6000h, which resulted in annual costs of 0,335 – 0,375 €/MWh$_{th}$ depending on the plant size. In general, the total annual costs increase as the operating hours decrease, and smaller plant size results in more rapid

increase in annual costs. As the operating hours increase, the cost curves approach each other. When the operating hours are as high as 7000h, the annual costs of different plant sizes are almost equal. This applies to cases where the share of capital costs is low. More capital intensive investments are more sensitive to costs variation within plant sizes.

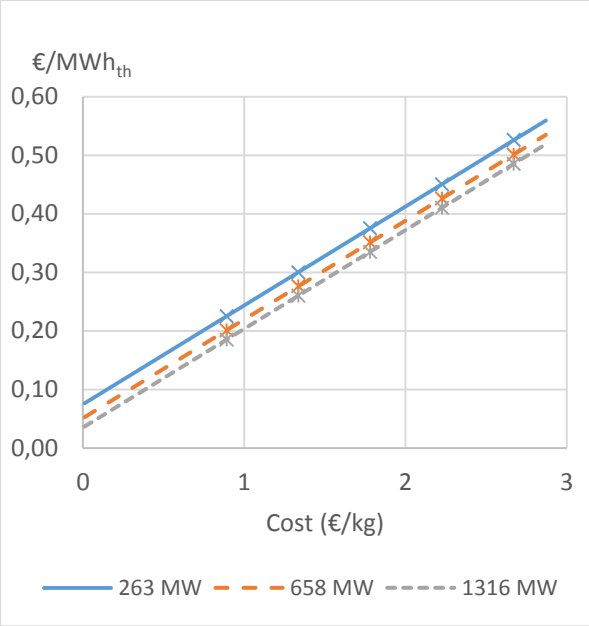

Figure 46: Impact of activated carbon price on annual costs in ESP plants with the emission level 1 µg/Nm^3.

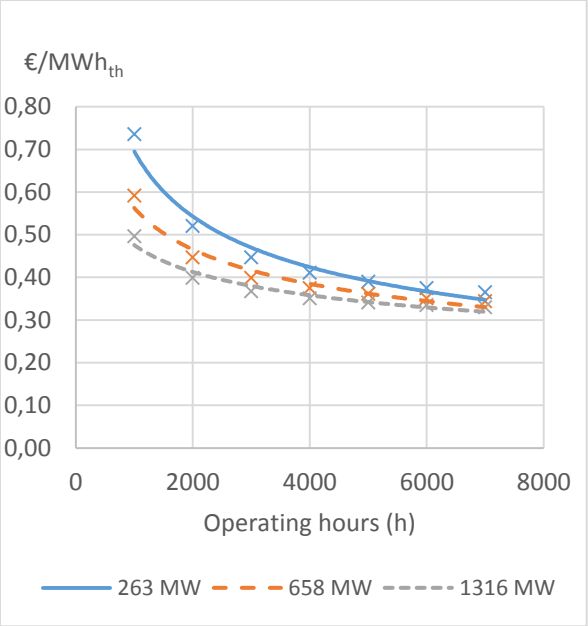

Figure 47: Impact of operating hours on annual costs in ESP plants with the emission level 1 µg/Nm^3.

Also, the impact of the mercury content of the coal on the annual costs and cost efficiency was studied. The results are shown in the Figure 48. The initial mercury content used in the study was 0,139 ppm, which resulted in annual costs of 0,335 – 0,375 €/MWh$_{th}$ or 23 800 – 26 600 €/kg of removed mercury, depending on the plant size. As seen in the Figure 48, the annual costs (€/MWh$_{th}$) increase when the mercury content of coal increases. This is because higher additional removal rate is required, which increases the activated carbon consumption. By contrast, the cost efficiency gets better when the mercury content increases. When mercury content is high, much more mercury in kilograms is removed without radical increase in operating costs, which improves the cost efficiency.

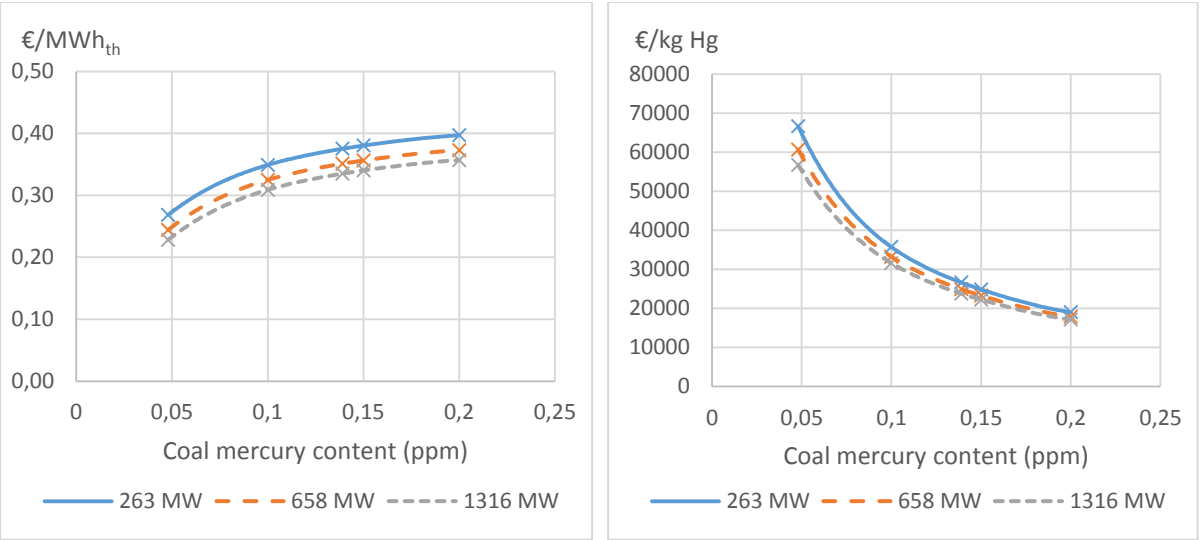

Figure 48: Impact of mercury content of coal on annual costs in ESP plants with the emission level 1 µg/Nm^3.

To investigate whether the operating hours have an influence on the competitiveness of different mercury removal technologies, some sensitivity analysis was made for plants that have ESP+WFGD configuration. A target emission level of 1 µg/Nm^3 was applied. The Figure 49 shows the annual costs and cost efficiency of untreated ACI, brominated ACI and bromine injection as a function of operating hours for the plant size 658 MW$_{th}$. As can be seen, operating hours have no effect on the competitiveness of different technologies. Actually the competitiveness of bromine injection seems to increase at low operating hours, probably due to lower investment costs. Same results were obtained for plant sizes 263 MW$_{th}$ and 1316 MW$_{th}$.

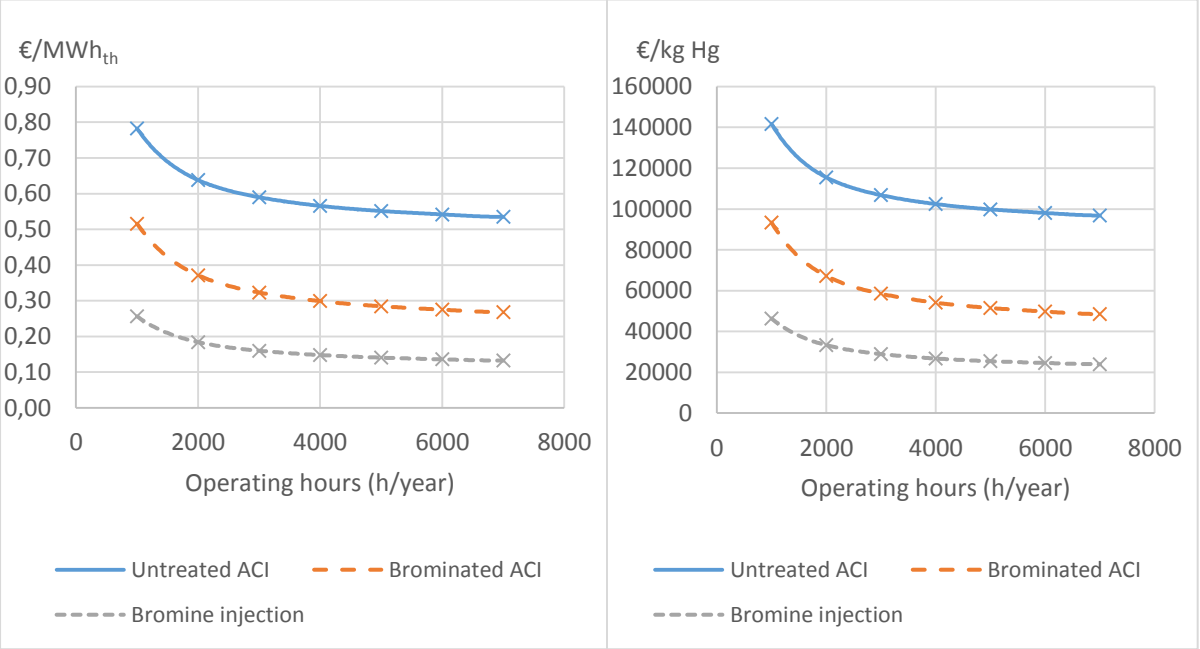

Figure 49: Annual costs and cost efficiency of mercury removal technologies in ESP+WFGD plants as a function of operating hours (plant capacity 658 MW$_{th}$ and emission level 1 µg/Nm^3).

10.3 Conclusions

First of all, it became clear that significant amounts of mercury can be removed in existing air pollution control devices as a co-benefit effect, if bituminous coal is used. In some cases this baseline removal efficiency of the existing equipment is so high that additional mercury removal is not needed to comply with the emission limits of the BREF document. Only plants with ESP alone need to cut their mercury emissions to get below 9 μg/Nm^3, which is the upper boundary for small plants. Besides ESP, plants with ESP+WFGD need to cut emissions when 4 μg/Nm^3 is pursued, which is the upper boundary for large plants. Furthermore, plant configurations FF, ESP+SDA+FF and SCR+ESP+WFGD need to cut emissions in order to get below 1 μg/Nm^3, which is the lowest boundary for all plant sizes. Plants with SCR+ESP+SDA+FF remove mercury so efficiently that the baseline mercury emissions are below 1 μg/Nm^3, so there is no need for additional mercury removal at all.

According to this cost analysis, bromine injection seems to be the most cost-efficient way to cut emissions in all plant sizes and emission levels. Thus, bromine injection should be preferred when possible, meaning configurations where desulphurization system is applied. In plants, where bromine injection is not applicable, brominated activated carbon injection could be the best alternative, having lower costs than untreated activated carbon injection.

In general, if plants need to comply with stricter emission limits, costs are relatively low and only a small increase in the electricity price would occur. The maximum annual costs to reduce mercury emissions below the levels 9, 4 and 1 μg/Nm^3 were calculated to be 0,153 €/MWh$_{th}$, 0,410 €/MWh$_{th}$ and 0,697 €/MWh$_{th}$, respectively. The corresponding increases in electricity prices would be 0,040, 0,108 and 0,183 cent/kWh$_e$. These prices would occur for 263 MW$_{th}$ plants with ESP only and that apply untreated ACI. In the worst case scenario, there would be some increase in electricity price, but it remains relatively low. For a large electrically heated household with a yearly consumption of 20000 kWh, an increment of 0,183 cent/kWh$_e$ would mean 36,6 € more costs per year, if additional charges are excluded.

I was noticed that configurations with fabric filter and SCR may be very efficient in mercury capture. These were excluded from this analysis, because they are not used directly for mercury removal and cannot be regarded as mercury-specific removal technologies. However, in collaboration with other pollutant removal targets, an installation of fabric filter or SCR could also be a cost efficient way to reduce mercury emissions. In particular, the TRAC-Catalyst introduced in the chapter 6.2.2 might be a good solution, especially for coal with a low halogen content. These should be considered as an alternative for the mercury-specific technologies that were included in this study.

It was also noticed that in some cases, the low additional removal requirement led to poor cost efficiency. For example, the required additional removal rate for ESP+SDA+FF configuration was only 16,6%, which led to 374 400 €/kg Hg removal costs in small plants. The baseline emissions of ESP+SDA+FF configuration was only 1,2 μg/Nm^3, exceeding the strictest emission limit by only 0,2 μg/Nm^3. Firstly, a deviation that small is hardly measurable and could be within error margins of the measuring equipment. Furthermore, it might not be reasonable to install a mercury removal system to reduce emissions by 16,6%, because the system would run

at a very low operating load, which will result in high costs. Instead, less expensive measures could be taken to fine-tune the emissions below 1 μg/Nm^3. This kind of measures include enhancement of existing flue gas cleaning devices and overall improvement of the power plant equipment.

Consequently, it is likely that mercury removal technologies will be implemented in plants that have a real need for additional removal. It is not very realistic nor economical to implement such technologies in plants that only slightly exceed the applied emission level, because it will lead to low cost-efficiency. In this study, many of the cases required only a slight decrease in mercury emissions to get below the applied emission levels, which were 9, 4 and 1 μg/Nm^3. As mentioned before, the emission limit for a plant is up to national or local authorities, and may vary plant by plant. This will make the decision making even more complicated, because the need for additional removal depends on plant-specific emission limits.

It should be remembered that the cost estimates of this analysis represent study level approximations. Thus the accuracy is low and the estimates will inevitably fall in the cost estimate class 5 (accuracy range -50 – 100%), which was introduced in the Table 22. In reality, there are many factors that affect the mercury removal costs. These are taxes, contingencies, economic life of capital equipment, process disruptions, price and performance of the consumables (sorbents and halogens), modifications to the existing equipment etc. The cost-estimates of this analysis are based on the assumptions that the retrofitting of mercury reduction systems is uncomplicated and no economic impact will occur due to installation. The mercury reduction system installation is assumed to be occurred in schedule during planned outages. Furthermore, no harmful impacts are estimated to happen to the existing equipment due to new installations.

Besides of the cost information, the highest uncertainties of this analysis are baseline mercury removal of existing air pollution control equipment and the consumption of sorbent/halogen. As mentioned before, these factors are very plant-specific, and in order to get reliable results, on-site measurements should be made. In this analysis, a generalized basis had to be taken, and the baseline removal efficiency and sorbent consumption were estimated by the basis of field test results in various other power plants.

Because the reference coal corresponded bituminous coal, high baseline removal efficiencies were achieved in the cost analysis. This in turn decreased the costs of additional removal technologies. As can be seen in the Table 20, the removal efficiencies of subbituminous coal and lignite are much lower. Thus, the costs for other lower rank coal types are probably much higher. For the future purposes, a similar analysis could be made for other coal types that have different baseline mercury removal efficiency.

To summarize, the costs of mercury removal represented here are based on various assumptions and conditions. Because each plant has its own requirements and challenges, the costs can vary significantly. The intention of this analysis was to give an overview of the total mercury removal costs and therefore the results cannot be taken as actual expected costs. Instead, the results give an indication of the general range of the costs that may occur when plants have to cut their mercury emissions below the levels set by the BREF document of the IED.

11 Summary

In this study, different mercury emission control technologies were reviewed and a cost analysis was made for some of the most common ways to reduce mercury emission. The costs were calculated in the context of mercury emission limits set by the Best Available Technologies Reference (BREF)-document that is related to the Industrial Emissions Directive (IED). The main objective of this study was to determine the costs of different mercury emission control technologies, when the emission limits of BREF document are applied. Cost analysis was made for existing plants with different flue gas cleaning configurations. The goal was to figure out whether the existing equipment can provide high enough mercury removal, and if not, what are the applicable mercury-specific removal technologies, and the costs caused by the additional mercury removal.

The analysis involved the following steps: determining the baseline mercury emissions and baseline mercury removal rate across the existing flue gas cleaning devices, calculating the additional required removal rates needed to achieve the BREF emission limits, and finally estimating the costs of different mercury-specific technologies that would occur if those emission limits were achieved.

Three mercury specific technologies were included in the analysis: activated carbon injection, brominated activated carbon injection and bromine salt injection. The analysis was made for existing coal-fired power plants that already applied some kind of flue gas cleaning devices. Calculations were made for three different power plant capacity, which were 100, 250 and 500 MW_e of electrical power, corresponding thermal powers of 263, 658 and 1316 $MW_{th.}$. Six different flue gas cleaning configurations were involved, which were ESP, FF, ESP+WFGD, ESP+SDA+FF, SCR+ESP+WFGD and SCR+ESP+SDA+FF. The costs were calculated for three different emission levels (BAT-AELs) that were 9, 4 and 1 µg/Nm³.

As the mercury removal efficiency of existing flue gas cleaning equipment depends strongly on the fuel properties, it was important to determine the fuel type that was used in the cost analysis. The properties of the reference coal that was used in the calculations were based on coal analyses for Polish coal. The reference coal corresponded bituminous hard coal that had high chlorine and low sulphur content, and a mercury content of 0,139 ppm. This led to baseline mercury emissions of 17,14 µg/Nm^3 (6% O_2, dry). The baseline mercury removal rates of the existing flue gas cleaning equipment were based on data that was gathered from various literature sources.

The baseline mercury removal efficiency of the studied plant configurations was discovered to be very good in general, when bituminous hard coal is used. In some cases the baseline removal efficiency was sufficient to achieve the emission limits of the BREF document without any additional mercury removal technology. The lowest baseline removal rate of 30% was determined for plants with only ESP, which led to baseline emissions of 12,0 µg/Nm^3. Therefore, additional removal technology was needed to get below the highest limit of 9 µg/Nm^3. The highest baseline removal efficiency of 98 % was determined for configuration SCR+ESP+SDA+FF, which led to baseline emissions of 0,3 µg/Nm^3. This meant that even the

strictest emission limit of 1 μg/Nm^3 was achieved without additional mercury removal technology.

Investment costs, operating costs and consumption values were determined by literature sources and interviews with technology suppliers. Investment costs were annualized by an interest rate of 8% and investment lifetime of 15 years. The cost evaluation within the studied technologies was made by comparing the annual costs (€/MWh_{th}) and cost efficiency (€/kg Hg removed). Yearly operating hours of the studied plants were determined to be 6000h. The consumption of the activated carbon was approximated by an algorithm developed by US Environment Protection Agency (US EPA). The algorithm was developed by analysing the data from field tests of several US power plants. The consumption rate of bromine is based on literature data that has been obtained in field tests.

Finally, the costs were calculated for each mercury removal technology. It became clear that in general, the annual costs caused by mercury removal are relatively low, varying between 0,034 – 0,697 €/MWh_{th}. The cost efficiency varies between 18 000 – 374 400 €/kg mercury removed. The increase in electricity price would be 0,009 - 0,183 cents/kWh. The costs depend on the baseline mercury removal rate, plant size, allowed mercury emission level and applied mercury-specific removal technology.

The cheapest technology seems to be bromine injection for all plant sizes, emission levels and operating hours. However, the applicability of bromine injection is limited to plants with a desulphurization system. For plants that cannot apply bromine injection, brominated activated carbon is preferred. Brominated activated carbon was noticed to be cheaper than untreated activated carbon for all plant sizes, emission levels and operating hours. At low additional mercury removal rates, the competitiveness of brominated activated carbon decreases in comparison to untreated activated carbon.

To compare the technologies, the costs in ESP+WFGD plants with emission level of 4μg/Nm^3 were 0,11 – 0,15 €/MWh_{th} for untreated activated carbon injection, 0,08 – 0,12 €/MWh_{th} for brominated activated carbon injection and 0,03 – 0,06 €/MWh_{th} for bromine injection. The cost efficiencies were 66 500 – 90 400 €/kg Hg for untreated activated carbon injection, 48 200 – 72 100 €/kg Hg for brominated activated carbon injection and 20 500 – 35 800 €/kg Hg for bromine injection.

It has to be mentioned that most of the data used in the analysis originate from pilot-scale plant tests. The results of the analysis are study-level approximations and too generalized conclusions should not be drawn. The costs in an individual plant depend on fuel properties, existing air pollution control equipment, boiler conditions, flue gas temperature and other plant-specific characteristics. However, the study gives some idea about the costs of different removal technologies for different cases, if they need to comply with the mercury emission limits of the BREF document.

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Appendices

Appendix 1. Mercury mass flows of the studied flue gas cleaning configurations. (3 pages)
Appendix 2. Calculations for untreated ACI. (5 pages)
Appendix 3. Calculations for brominated ACI. (5 pages)
Appendix 4. Calculations for bromine injection. (3 pages)

Appendix 1: Mercury mass flows of the studied flue gas cleaning configurations

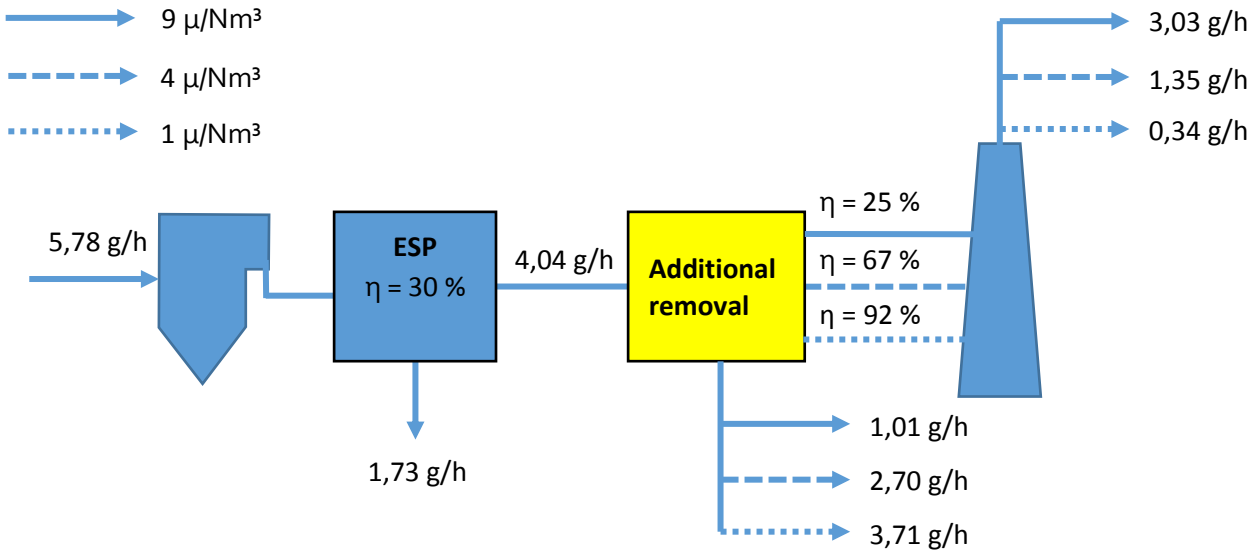

Figure 0.1: Mercury mass flows for plants with ESP. Mass flows are calculated for plant size 263 MW_{th} for emission limits 9, 4 and 1 µ/Nm³.

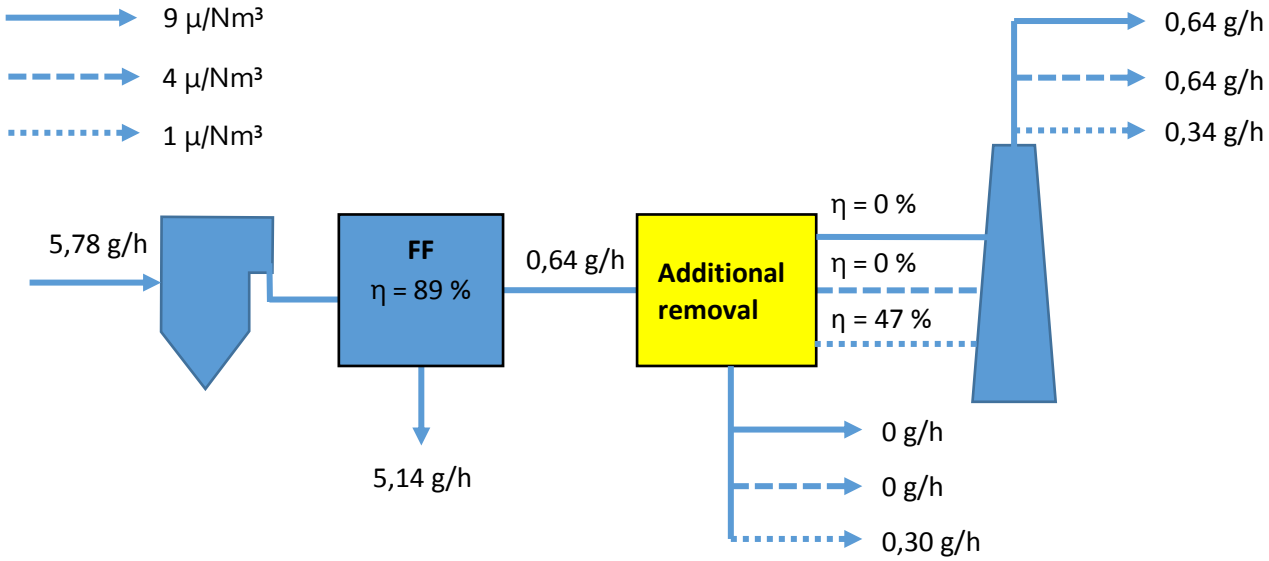

Figure 0.2: Mercury mass flows for plants with FF. Mass flows are calculated for plant size 263 MW_{th} for emission limits 9, 4 and 1 µ/Nm³.

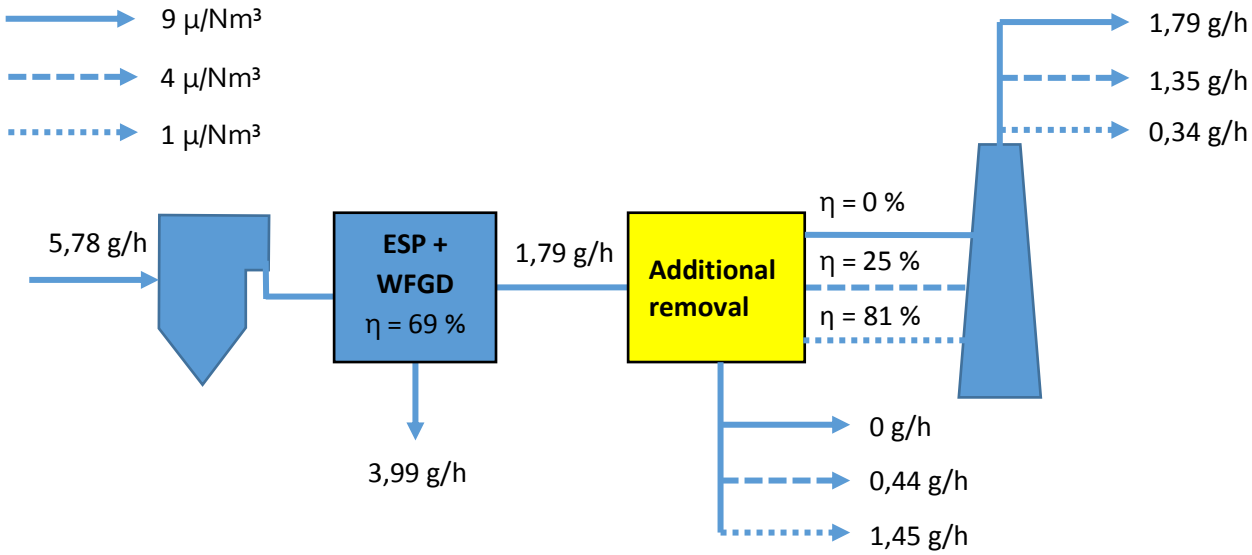

Figure 0.3: Mercury mass flows for plants with ESP + WFGD. Mass flows are calculated for plant size 263 MW_{th} for emission limits 9, 4 and 1 µ/Nm³.

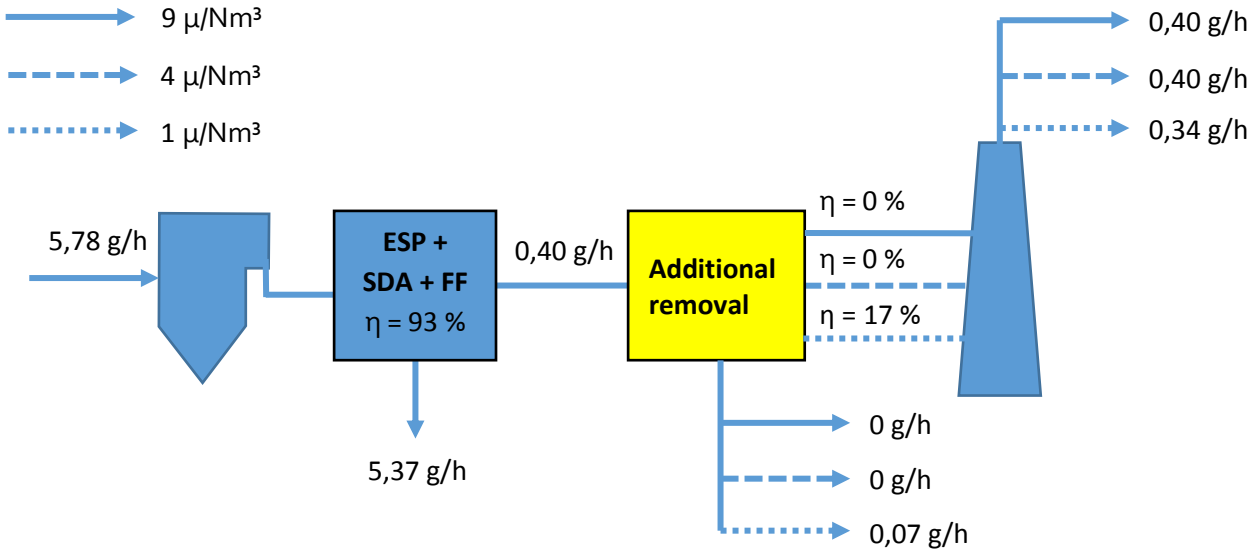

Figure 0.4: Mercury mass flows for plants with ESP + SDA + FF. Mass flows are calculated for plant size 263 MW_{th} for emission limits 9, 4 and 1 µ/Nm³.

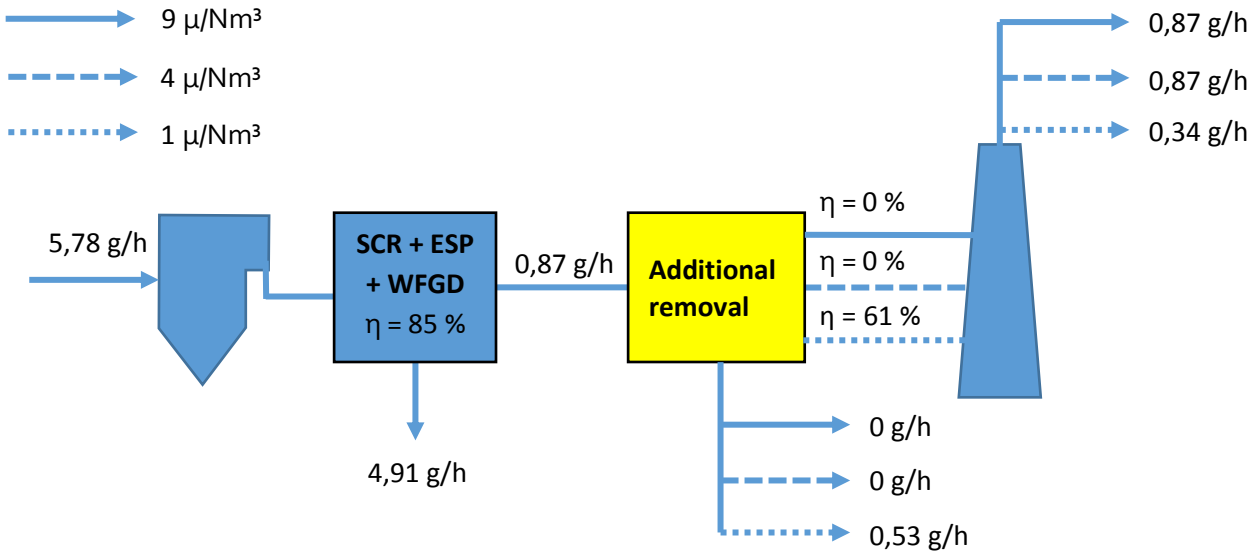

Figure 0.5: Mercury mass flows for plants with SCR + ESP + WFGD. Mass flows are calculated for plant size 263 MW_{th} for emission limits 9, 4 and 1 µ/Nm³.

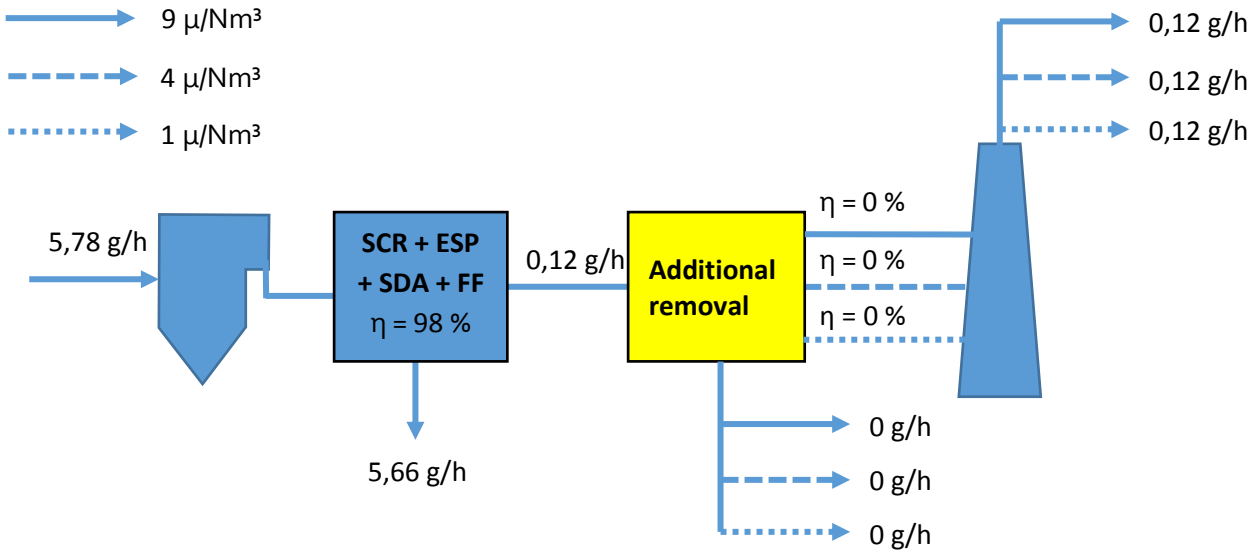

Figure 0.6: Mercury mass flows for plants with SCR + ESP + SDA + FF. Mass flows are calculated for plant size 263 MW_{th} for emission limits 9, 4 and 1 µ/Nm³.

Appendix 2: Calculations for untreated ACI

Existing flue gas cleaning configuration: ESP

	Emission limit: 9 µg/Nm3			Emission limit: 4 µg/Nm3			Emission limit: 1 µg/Nm3		
MERCURY EMISSIONS CALCULATION	**263 MW**	**658 MW**	**1316 MW**	**263 MW**	**658 MW**	**1316 MW**	**263 MW**	**658 MW**	**1316 MW**
POWER PLANT PROPERTIES									
Electrical power (MW)	100	250	500	100	250	500	100	250	500
Thermal power (MW)	263	658	1316	263	658	1316	263	658	1316
Fuel consumption (t/a)	249307	623269	1246537	249307	623269	1246537	249307	623269	1246537
FLUE GAS									
Flue gas volume-to-mass ratio (Nm3/kg fuel) (dry)	8,507	8,507	8,507	8,507	8,507	8,507	8,507	8,507	8,507
Flue gas volume flow (Nm3/h) (dry)	353476	883691	1767382	353476	883691	1767382	353476	883691	1767382
MERCURY REMOVAL									
MERCURY INPUT									
Hg input (kg/a)	34,65	86,63	173,27	34,65	86,63	173,27	34,65	86,63	173,27
Hg input (µg/Nm3) (dry, 6% O2)	17,14	17,14	17,14	17,14	17,14	17,14	17,14	17,14	17,14
MERCURY REMOVAL IN EXISTING EQUIPMENT									
Baseline Hg removal efficiency (%)	30,0 %	30,0 %	30,0 %	30,0 %	30,0 %	30,0 %	30,0 %	30,0 %	30,0 %
Baseline Hg removal (kg/a)	10,40	25,99	51,98	10,40	25,99	51,98	10,40	25,99	51,98
Baseline Hg removal (µg/Nm3) (dry, 6% O2)	5,14	5,14	5,14	5,14	5,14	5,14	5,14	5,14	5,14
Baseline Hg emissions (kg/a)	24,26	60,64	121,29	24,26	60,64	121,29	24,26	60,64	121,29
Baseline Hg emissions (µg/Nm3) (dry, 6% O2)	12,0	12,0	12,0	12,0	12,0	12,0	12,0	12,0	12,0
Hg emission limit (µg/Nm3)	9	9	9	4	4	4	1	1	1
ADDITIONAL MERCURY REMOVAL									
Required additional removal (%)	25,0 %	25,0 %	25,0 %	66,7 %	66,7 %	66,7 %	91,7 %	91,7 %	91,7 %
Required additional removal (kg/a)	6,06	15,15	30,30	16,17	40,43	80,85	22,24	55,59	111,18
Required additional removal (µg/Nm3) (dry, 6% O2)	3,0	3,0	3,0	8,0	8,0	8,0	11,0	11,0	11,0
TOTAL MERCURY REMOVAL									
Total Hg removal (%)	47,5 %	47,5 %	47,5 %	76,7 %	76,7 %	76,7 %	94,2 %	94,2 %	94,2 %
Total Hg removal (kg/a)	16,46	41,14	82,28	26,57	66,42	132,83	32,63	81,58	163,16
Total Hg removal (µg/Nm3) (dry, 6% O2)	8,14	8,14	8,14	13,14	13,14	13,14	16,14	16,14	16,14
MERCURY STACK EMISSIONS									
Hg stack emissions (kg/a)	18,20	45,49	90,98	8,09	20,22	40,44	2,02	5,05	10,11
Hg stack emissions (µg/Nm3) (dry,	9,0	9,0	9,0	4,0	4,0	4,0	1,0	1,0	1,0
COST-ANALYSIS	**263 MW**	**658 MW**	**1316 MW**	**263 MW**	**658 MW**	**1316 MW**	**263 MW**	**658 MW**	**1316 MW**
INVESTMENT COSTS									
Total investment cost (€/kWe)	8,01	5,34	3,56	8,01	5,34	3,56	8,01	5,34	3,56
Total investment cost (€)	**801 000**	**1 335 000**	**1 780 000**	**801 000**	**1 335 000**	**1 780 000**	**801 000**	**1 335 000**	**1 780 000**
CONSUMPTION RATES									
Consumption of the sorbent (mg/Nm3)	42,4	42,4	42,4	181,2	181,2	181,2	335,7	335,7	335,7
Consumption of the sorbent (kg/h)	15,6	38,9	77,8	66,5	166,2	332,5	123,2	308,0	616,0
Electricity (kW)	6,50	16,20	32,40	6,50	16,20	32,40	6,50	16,20	32,40
Electricity (MWh/a)	39,00	97,20	194,40	39,00	97,20	194,40	39,00	97,20	194,40
OPERATING COSTS									
Variable operating costs									
Sorbent (€/a)	124 169	310 422	620 843	530 645	1 326 613	2 653 226	983 099	2 457 748	4 915 496
Electricity (€/a)	3 171	7 902	15 805	3 171	7 902	15 805	3 171	7 902	15 805
Total variable costs (€/a)	**127 339**	**318 324**	**636 648**	**533 816**	**1 334 515**	**2 669 030**	**986 270**	**2 465 650**	**4 931 301**
Fixed operating costs									
Operation and maintenance (%/investment cost)	3 %	3 %	3 %	3 %	3 %	3 %	3 %	3 %	3 %
Operation and maintenance (€/a)	24 030	40 050	53 400	24 030	40 050	53 400	24 030	40 050	53 400
Total fixed costs (€/a)	**24 030**	**40 050**	**53 400**	**24 030**	**40 050**	**53 400**	**24 030**	**40 050**	**53 400**
Total operating costs (€/a)	**151 369**	**358 374**	**690 048**	**557 846**	**1 374 565**	**2 722 430**	**1 010 300**	**2 505 700**	**4 984 701**
YEARLY COSTS									
Interest rate (%)	8 %	8 %	8 %	8 %	8 %	8 %	8 %	8 %	8 %
Lifetime of the investment (a)	15	15	15	15	15	15	15	15	15
Annual investment costs (€/a)	90048	150080	200106	90048	150080	200106	90048	150080	200106
Operating costs (€/a)	151369	358374	690048	557846	1374565	2722430	1010300	2505700	4984701
TOTA YEARLY COSTS (€/a)	**241417**	**508454**	**890154**	**647894**	**1524645**	**2922537**	**1100348**	**2655780**	**5184807**
NOMINAL COSTS									
Annual investment cost (€/MWhth)	0,057	0,038	0,025	0,057	0,038	0,025	0,057	0,038	0,025
Variable costs (€/MWhth)	0,081	0,081	0,081	0,338	0,338	0,338	0,625	0,625	0,625
Fixed costs (€/MWhth)	0,015	0,010	0,007	0,015	0,010	0,007	0,015	0,010	0,007
Annual costs (€/MWhth)	0,153	0,129	0,113	0,410	0,386	0,370	0,697	0,673	0,657
Increase in electricity price (cent/kWhe)	0,040	0,034	0,030	0,108	0,102	0,097	0,183	0,177	0,173
Cost effciency (€/kgHg)	39834	33558	29375	40067	37715	36147	49486	47775	46635

Existing flue gas cleaning configuration: FF

	Emission limit: 1 µg/Nm3		
MERCURY EMISSIONS CALCULATION	**263 MW**	**658 MW**	**1316 MW**
POWER PLANT PROPERTIES			
Electrical power (MW)	100	250	500
Thermal power (MW)	263	658	1316
Fuel consumption (t/a)	249307	623269	1246537
FLUE GAS			
Flue gas volume-to-mass ratio (Nm3/kg fuel) (dry)	8,507	8,507	8,507
Flue gas volume flow (Nm3/h) (dry)	353476	883691	1767382
MERCURY REMOVAL			
MERCURY INPUT			
Hg input (kg/a)	34,65	86,63	173,27
Hg input (µg/Nm3) (dry, 6% O2)	17,14	17,14	17,14
MERCURY REMOVAL IN EXISTING EQUIPMENT			
Baseline Hg removal efficiency (%)	89,0 %	89,0 %	89,0 %
Baseline Hg removal (kg/a)	30,84	77,10	154,21
Baseline Hg removal (µg/Nm3) (dry, 6% O2)	15,25	15,25	15,25
Baseline Hg emissions (kg/a)	3,81	9,53	19,06
Baseline Hg emissions (µg/Nm3) (dry, 6% O2)	1,9	1,9	1,9
Hg emission limit (µg/Nm3)	1	1	1
ADDITIONAL MERCURY REMOVAL			
Required additional removal (%)	47,0 %	47,0 %	47,0 %
Required additional removal (kg/a)	1,79	4,48	8,95
Required additional removal (µg/Nm3) (dry, 6% O2)	0,9	0,9	0,9
TOTAL MERCURY REMOVAL			
Total Hg removal (%)	94,2 %	94,2 %	94,2 %
Total Hg removal (kg/a)	32,63	81,58	163,16
Total Hg removal (µg/Nm3) (dry, 6% O2)	16,14	16,14	16,14
MERCURY STACK EMISSIONS			
Hg stack emissions (kg/a)	2,02	5,05	10,11
Hg stack emissions (µg/Nm3) (dry,	1,0	1,0	1,0
COST-ANALYSIS	**263 MW**	**658 MW**	**1316 MW**
INVESTMENT COSTS			
Total investment cost (€/kWe)	8,01	5,34	3,56
Total investment cost (€)	**801 000**	**1 335 000**	**1 780 000**
CONSUMPTION RATES			
Consumption of the sorbent (mg/Nm3)	11,2	11,2	11,2
Consumption of the sorbent (kg/h)	4,1	10,3	20,6
Electricity (kW)	6,50	16,20	32,40
Electricity (MWh/a)	39,00	97,20	194,40
OPERATING COSTS			
Variable operating costs			
Sorbent (€/a)	32 799	81 998	163 996
Electricity (€/a)	3 171	7 902	15 805
Total variable costs (€/a)	**35 970**	**89 901**	**179 801**
Fixed operating costs			
Operation and maintenance (%/investment cost)	3 %	3 %	3 %
Operation and maintenance (€/a)	24 030	40 050	53 400
Total fixed costs (€/a)	**24 030**	**40 050**	**53 400**
Total operating costs (€/a)	**60 000**	**129 951**	**233 201**
YEARLY COSTS			
Interest rate (%)	8 %	8 %	8 %
Lifetime of the investment (a)	15	15	15
Annual investment costs (€/a)	90048	150080	200106
Operating costs (€/a)	60000	129951	233201
TOTA YEARLY COSTS (€/a)	**150048**	**280030**	**433307**
NOMINAL COSTS			
Annual investment cost (€/MWhth)	0,057	0,038	0,025
Variable costs (€/MWhth)	0,023	0,023	0,023
Fixed costs (€/MWhth)	0,015	0,010	0,007
Annual costs (€/MWhth)	0,095	0,071	0,055
Increase in electricity price (cent/kWhe)	0,025	0,019	0,014
Cost effciency (€/kgHg)	83824	62576	48414

Existing flue gas cleaning configuration: ESP+WFGD

	Emission limit: 4 µg/Nm3			Emission limit: 1 µg/Nm3		
MERCURY EMISSIONS CALCULATION	**263 MW**	**658 MW**	**1316 MW**	**263 MW**	**658 MW**	**1316 MW**
POWER PLANT PROPERTIES						
Electrical power (MW)	100	250	500	100	250	500
Thermal power (MW)	263	658	1316	263	658	1316
Fuel consumption (t/a)	249307	623269	1246537	249307	623269	1246537
FLUE GAS						
Flue gas volume-to-mass ratio (Nm3/kg fuel) (dry)	8,507	8,507	8,507	8,507	8,507	8,507
Flue gas volume flow (Nm3/h) (dry)	353476	883691	1767382	353476	883691	1767382
MERCURY REMOVAL						
MERCURY INPUT						
Hg input (kg/a)	34,65	86,63	173,27	34,65	86,63	173,27
Hg input (µg/Nm3) (dry, 6% O2)	17,14	17,14	17,14	17,14	17,14	17,14
MERCURY REMOVAL IN EXISTING EQUIPMENT						
Baseline Hg removal efficiency (%)	69,0 %	69,0 %	69,0 %	69,0 %	69,0 %	69,0 %
Baseline Hg removal (kg/a)	23,91	59,78	119,56	23,91	59,78	119,56
Baseline Hg removal (µg/Nm3) (dry, 6% O2)	11,83	11,83	11,83	11,83	11,83	11,83
Baseline Hg emissions (kg/a)	1,79	4,48	8,95	1,79	4,48	8,95
Baseline Hg emissions (µg/Nm3) (dry, 6% O2)	5,3	5,3	5,3	5,3	5,3	5,3
Hg emission limit (µg/Nm3)	4	4	4	1	1	1
ADDITIONAL MERCURY REMOVAL						
Required additional removal (%)	24,7 %	24,7 %	24,7 %	81,2 %	81,2 %	81,2 %
Required additional removal (kg/a)	2,66	6,64	13,28	8,72	21,80	43,60
Required additional removal (µg/Nm3) (dry, 6% O2)	1,3	1,3	1,3	4,3	4,3	4,3
TOTAL MERCURY REMOVAL						
Total Hg removal (%)	76,7 %	76,7 %	76,7 %	94,2 %	94,2 %	94,2 %
Total Hg removal (kg/a)	26,57	66,42	132,83	32,63	81,58	163,16
Total Hg removal (µg/Nm3) (dry, 6% O2)	13,14	13,14	13,14	16,14	16,14	16,14
MERCURY STACK EMISSIONS						
Hg stack emissions (kg/a)	8,09	20,22	40,44	2,02	5,05	10,11
Hg stack emissions (µg/Nm3) (dry, 6% O2)	4,0	4,0	4,0	1,0	1,0	1,0
COST-ANALYSIS	**263 MW**	**658 MW**	**1316 MW**	**263 MW**	**658 MW**	**1316 MW**
INVESTMENT COSTS						
Total investment cost (€/kWe)	8,01	5,34	3,56	8,01	5,34	3,56
Total investment cost (€)	**801 000**	**1 335 000**	**1 780 000**	**801 000**	**1 335 000**	**1 780 000**
CONSUMPTION RATES						
Consumption of the sorbent (mg/Nm3)	41,9	41,9	41,9	265	265	265
Consumption of the sorbent (kg/h)	15,4	38,4	76,9	97,2	243,1	486,2
Electricity (kW)	6,50	16,20	32,40	6,50	16,20	32,40
Electricity (MWh/a)	39,00	97,20	194,40	39,00	97,20	194,40
OPERATING COSTS						
Variable operating costs						
Sorbent (€/a)	122 704	306 761	613 522	776 054	1 940 135	3 880 269
Electricity (€/a)	3 171	7 902	15 805	3 171	7 902	15 805
Total variable costs (€/a)	**125 875**	**314 663**	**629 327**	**779 225**	**1 948 037**	**3 896 074**
Fixed operating costs						
Operation and maintenance (%/investment cost)	3 %	3 %	3 %	3 %	3 %	3 %
Operation and maintenance (€/a)	24 030	40 050	53 400	24 030	40 050	53 400
Total fixed costs (€/a)	**24 030**	**40 050**	**53 400**	**24 030**	**40 050**	**53 400**
Total operating costs (€/a)	**149 905**	**354 713**	**682 727**	**803 255**	**1 988 087**	**3 949 474**
YEARLY COSTS						
Interest rate (%)	8 %	8 %	8 %	8 %	8 %	8 %
Lifetime of the investment (a)	15	15	15	15	15	15
Annual investment costs (€/a)	90048	150080	200106	90048	150080	200106
Operating costs (€/a)	149905	354713	682727	803255	1988087	3949474
TOTA YEARLY COSTS (€/a)	**239953**	**504793**	**882833**	**893302**	**2138167**	**4149581**
NOMINAL COSTS						
Annual investment cost (€/MWhth)	0,057	0,038	0,025	0,057	0,038	0,025
Variable costs (€/MWhth)	0,080	0,080	0,080	0,494	0,494	0,494
Fixed costs (€/MWhth)	0,015	0,010	0,007	0,015	0,010	0,007
Annual costs (€/MWhth)	0,152	0,128	0,112	0,566	0,542	0,526
Increase in electricity price (cent/kWhe)	0,040	0,034	0,029	0,149	0,143	0,138
Cost effciency (€/kgHg)	90374	76048	66500	102434	98072	95165

Existing flue gas cleaning configuration: ESP+SDA+FF

	Emission limit: 1 µg/Nm3		
MERCURY EMISSIONS CALCULATION	**263 MW**	**658 MW**	**1316 MW**
POWER PLANT PROPERTIES			
Electrical power (MW)	100	250	500
Thermal power (MW)	263	658	1316
Fuel consumption (t/a)	249307	623269	1246537
FLUE GAS			
Flue gas volume-to-mass ratio (Nm3/kg fuel) (dry)	8,507	8,507	8,507
Flue gas volume flow (Nm3/h) (dry)	353476	883691	1767382
MERCURY REMOVAL			
MERCURY INPUT			
Hg input (kg/a)	34,65	86,63	173,27
Hg input (µg/Nm3) (dry, 6% O2)	17,14	17,14	17,14
MERCURY REMOVAL IN EXISTING EQUIPMENT			
Baseline Hg removal efficiency (%)	93,0 %	93,0 %	93,0 %
Baseline Hg removal (kg/a)	32,23	80,57	161,14
Baseline Hg removal (µg/Nm3) (dry, 6% O2)	15,94	15,94	15,94
Baseline Hg emissions (kg/a)	2,43	6,06	12,13
Baseline Hg emissions (µg/Nm3) (dry, 6% O2)	1,2	1,2	1,2
Hg emission limit (µg/Nm3)	1	1	1
ADDITIONAL MERCURY REMOVAL			
Required additional removal (%)	16,6 %	16,6 %	16,6 %
Required additional removal (kg/a)	0,40	1,01	2,02
Required additional removal (µg/Nm3) (dry, 6% O2)	0,2	0,2	0,2
TOTAL MERCURY REMOVAL			
Total Hg removal (%)	94,2 %	94,2 %	94,2 %
Total Hg removal (kg/a)	32,63	81,58	163,16
Total Hg removal (µg/Nm3) (dry, 6% O2)	16,14	16,14	16,14
MERCURY STACK EMISSIONS			
Hg stack emissions (kg/a)	2,02	5,05	10,11
Hg stack emissions (µg/Nm3) (dry, 6% O2)	1,0	1,0	1,0
COST-ANALYSIS	**263 MW**	**658 MW**	**1316 MW**
INVESTMENT COSTS			
Total investment cost (€/kWe)	8,01	5,34	3,56
Total investment cost (€)	**801 000**	**1 335 000**	**1 780 000**
CONSUMPTION RATES			
Consumption of the sorbent (mg/Nm3)	11,6	11,6	11,6
Consumption of the sorbent (kg/h)	4,3	10,6	21,3
Electricity (kW)	6,50	16,20	32,40
Electricity (MWh/a)	39,00	97,20	194,40
OPERATING COSTS			
Variable operating costs			
Sorbent (€/a)	33 971	84 927	169 853
Electricity (€/a)	3 171	7 902	15 805
Total variable costs (€/a)	**37 141**	**92 829**	**185 658**
Fixed operating costs			
Operation and maintenance (%/investment cost)	3 %	3 %	3 %
Operation and maintenance (€/a)	24 030	40 050	53 400
Total fixed costs (€/a)	**24 030**	**40 050**	**53 400**
Total operating costs (€/a)	**61 171**	**132 879**	**239 058**
YEARLY COSTS			
Interest rate (%)	8 %	8 %	8 %
Lifetime of the investment (a)	15	15	15
Annual investment costs (€/a)	90048	150080	200106
Operating costs (€/a)	61171	132879	239058
TOTA YEARLY COSTS (€/a)	**151219**	**282959**	**439164**
NOMINAL COSTS			
Annual investment cost (€/MWhth)	0,057	0,038	0,025
Variable costs (€/MWhth)	0,024	0,024	0,024
Fixed costs (€/MWhth)	0,015	0,010	0,007
Annual costs (€/MWhth)	0,096	0,072	0,056
Increase in electricity price (cent/kWhe)	0,025	0,019	0,015
Cost effciency (€/kgHg)	374419	280243	217475

Existing flue gas cleaning configuration: SCR+ESP+WFGD

	Emission limit: 1 µg/Nm3		
MERCURY EMISSIONS CALCULATION	**263 MW**	**658 MW**	**1316 MW**
POWER PLANT PROPERTIES			
Electrical power (MW)	100	250	500
Thermal power (MW)	263	658	1316
Fuel consumption (t/a)	249307	623269	1246537
FLUE GAS			
Flue gas volume-to-mass ratio (Nm3/kg fuel) (dry)	8,507	8,507	8,507
Flue gas volume flow (Nm3/h) (dry)	353476	883691	1767382
MERCURY REMOVAL			
MERCURY INPUT			
Hg input (kg/a)	34,65	86,63	173,27
Hg input (µg/Nm3) (dry, 6% O2)	17,14	17,14	17,14
MERCURY REMOVAL IN EXISTING EQUIPMENT			
Baseline Hg removal efficiency (%)	85,0 %	85,0 %	85,0 %
Baseline Hg removal (kg/a)	29,46	73,64	147,28
Baseline Hg removal (µg/Nm3) (dry, 6% O2)	14,57	14,57	14,57
Baseline Hg emissions (kg/a)	5,20	13,00	25,99
Baseline Hg emissions (µg/Nm3) (dry, 6% O2)	2,6	2,6	2,6
Hg emission limit (µg/Nm3)	1	1	1
ADDITIONAL MERCURY REMOVAL			
Required additional removal (%)	61,1 %	61,1 %	61,1 %
Required additional removal (kg/a)	3,18	7,94	15,88
Required additional removal (µg/Nm3) (dry, 6% O2)	1,6	1,6	1,6
TOTAL MERCURY REMOVAL			
Total Hg removal (%)	94,2 %	94,2 %	94,2 %
Total Hg removal (kg/a)	32,63	81,58	163,16
Total Hg removal (µg/Nm3) (dry, 6% O2)	16,14	16,14	16,14
MERCURY STACK EMISSIONS			
Hg stack emissions (kg/a)	2,02	5,05	10,11
Hg stack emissions (µg/Nm3) (dry, 6% O2)	1,0	1,0	1,0
COST-ANALYSIS	**263 MW**	**658 MW**	**1316 MW**
INVESTMENT COSTS			
Total investment cost (€/kWe)	8,01	5,34	3,56
Total investment cost (€)	**801 000**	**1 335 000**	**1 780 000**
CONSUMPTION RATES			
Consumption of the sorbent (mg/Nm3)	154	154	154
Consumption of the sorbent (kg/h)	56,5	141,3	282,6
Electricity (kW)	6,50	16,20	32,40
Electricity (MWh/a)	39,00	97,20	194,40
OPERATING COSTS			
Variable operating costs			
Sorbent (€/a)	450 990	1 127 475	2 254 949
Electricity (€/a)	3 171	7 902	15 805
Total variable costs (€/a)	**454 161**	**1 135 377**	**2 270 754**
Fixed operating costs			
Operation and maintenance (%/investment cost)	3 %	3 %	3 %
Operation and maintenance (€/a)	24 030	40 050	53 400
Total fixed costs (€/a)	**24 030**	**40 050**	**53 400**
Total operating costs (€/a)	**478 191**	**1 175 427**	**2 324 154**
YEARLY COSTS			
Interest rate (%)	8 %	8 %	8 %
Lifetime of the investment (a)	15	15	15
Annual investment costs (€/a)	90048	150080	200106
Operating costs (€/a)	478191	1175427	2324154
TOTA YEARLY COSTS (€/a)	**568238**	**1325507**	**2524260**
NOMINAL COSTS			
Annual investment cost (€/MWhth)	0,057	0,038	0,025
Variable costs (€/MWhth)	0,288	0,288	0,288
Fixed costs (€/MWhth)	0,015	0,010	0,007
Annual costs (€/MWhth)	0,360	0,336	0,320
Increase in electricity price (cent/kWhe)	0,095	0,088	0,084
Cost effciency (€/kgHg)	178906	166931	158950

Appendix 3: Calculations for brominated ACI

Existing flue gas cleaning configuration: ESP

	Emission limit: 9 µg/Nm3			Emission limit: 4 µg/Nm3			Emission limit: 1 µg/Nm3		
MERCURY EMISSIONS CALCULATION	**263 MW**	**658 MW**	**1316 MW**	**263 MW**	**658 MW**	**1316 MW**	**263 MW**	**658 MW**	**1316 MW**
POWER PLANT PROPERTIES									
Electrical power (MW)	100	250	500	100	250	500	100	250	500
Thermal power (MW)	263	658	1316	263	658	1316	263	658	1316
Fuel consumption (t/a)	249307	623269	1246537	249307	623269	1246537	249307	623269	1246537
FLUE GAS									
Flue gas volume-to-mass ratio (Nm3/kg fuel) (dry)	8,507	8,507	8,507	8,507	8,507	8,507	8,507	8,507	8,507
Flue gas volume flow (Nm3/h) (dry)	353476	883691	1767382	353476	883691	1767382	353476	883691	1767382
MERCURY REMOVAL									
MERCURY INPUT									
Hg input (kg/a)	34,65	86,63	173,27	34,65	86,63	173,27	34,65	86,63	173,27
Hg input (µg/Nm3) (dry, 6% O2)	17,14	17,14	17,14	17,14	17,14	17,14	17,14	17,14	17,14
MERCURY REMOVAL IN EXISTING EQUIPMENT									
Baseline Hg removal efficiency (%)	30,0 %	30,0 %	30,0 %	30,0 %	30,0 %	30,0 %	30,0 %	30,0 %	30,0 %
Baseline Hg removal (kg/a)	10,40	25,99	51,98	10,40	25,99	51,98	10,40	25,99	51,98
Baseline Hg removal (µg/Nm3) (dry, 6% O2)	5,14	5,14	5,14	5,14	5,14	5,14	5,14	5,14	5,14
Baseline Hg emissions (kg/a)	24,26	60,64	121,29	24,26	60,64	121,29	24,26	60,64	121,29
Baseline Hg emissions (µg/Nm3) (dry, 6% O2)	12,0	12,0	12,0	12,0	12,0	12,0	12,0	12,0	12,0
Hg emission limit (µg/Nm3)	9	9	9	4	4	4	1	1	1
ADDITIONAL MERCURY REMOVAL									
Required additional removal (%)	25,0 %	25,0 %	25,0 %	66,7 %	66,7 %	66,7 %	91,7 %	91,7 %	91,7 %
Required additional removal (kg/a)	6,06	15,15	30,30	16,17	40,43	80,85	22,24	55,59	111,18
Required additional removal (µg/Nm3) (dry, 6% O2)	3,0	3,0	3,0	8,0	8,0	8,0	11,0	11,0	11,0
TOTAL MERCURY REMOVAL									
Total Hg removal (%)	47,5 %	47,5 %	47,5 %	76,7 %	76,7 %	76,7 %	94,2 %	94,2 %	94,2 %
Total Hg removal (kg/a)	16,46	41,14	82,28	26,57	66,42	132,83	32,63	81,58	163,16
Total Hg removal (µg/Nm3) (dry, 6% O2)	8,14	8,14	8,14	13,14	13,14	13,14	16,14	16,14	16,14
MERCURY STACK EMISSIONS									
Hg stack emissions (kg/a)	18,20	45,49	90,98	8,09	20,22	40,44	2,02	5,05	10,11
Hg stack emissions (µg/Nm3) (dry, 6% O2)	9,0	9,0	9,0	4,0	4,0	4,0	1,0	1,0	1,0
COST-ANALYSIS	**263 MW**	**658 MW**	**1316 MW**	**263 MW**	**658 MW**	**1316 MW**	**263 MW**	**658 MW**	**1316 MW**
INVESTMENT COSTS									
Total investment cost (€/kWe)	8,01	5,34	3,56	8,01	5,34	3,56	8,01	5,34	3,56
Total investment cost (€)	**801 000**	**1 335 000**	**1 780 000**	**801 000**	**1 335 000**	**1 780 000**	**801 000**	**1 335 000**	**1 780 000**
CONSUMPTION RATES									
Consumption of the sorbent (mg/Nm3)	19	19	19	60,5	60,5	60,5	121,2	121,2	121,2
Consumption of the sorbent (kg/h)	7,0	17,4	34,9	22,2	55,5	111,0	44,5	111,2	222,4
Electricity (kW)	6,50	16,20	32,40	6,50	16,20	32,40	6,50	16,20	32,40
Electricity (MWh/a)	39,00	97,20	194,40	39,00	97,20	194,40	39,00	97,20	194,40
OPERATING COSTS									
Variable operating costs									
Sorbent (€/a)	74 468	186 169	372 339	237 121	592 802	1 185 604	475 026	1 187 564	2 375 128
Electricity (€/a)	3 171	7 902	15 805	3 171	7 902	15 805	3 171	7 902	15 805
Total variable costs (€/a)	**77 638**	**194 072**	**388 143**	**240 292**	**600 704**	**1 201 409**	**478 196**	**1 195 466**	**2 390 933**
Fixed operating costs									
Operation and maintenance (%/investment cost)	3 %	3 %	3 %	3 %	3 %	3 %	3 %	3 %	3 %
Operation and maintenance (€/a)	24 030	40 050	53 400	24 030	40 050	53 400	24 030	40 050	53 400
Total fixed costs (€/a)	**24 030**	**40 050**	**53 400**	**24 030**	**40 050**	**53 400**	**24 030**	**40 050**	**53 400**
Total operating costs (€/a)	**101 668**	**234 122**	**441 543**	**264 322**	**640 754**	**1 254 809**	**502 226**	**1 235 516**	**2 444 333**
YEARLY COSTS									
Interest rate (%)	8 %	8 %	8 %	8 %	8 %	8 %	8 %	8 %	8 %
Lifetime of the investment (a)	15	15	15	15	15	15	15	15	15
Annual investment costs (€/a)	90048	150080	200106	90048	150080	200106	90048	150080	200106
Operating costs (€/a)	101668	234122	441543	264322	640754	1254809	502226	1235516	2444333
TOTA YEARLY COSTS (€/a)	**191716**	**384201**	**641650**	**354369**	**790834**	**1454915**	**592274**	**1385596**	**2644439**
NOMINAL COSTS									
Annual investment cost (€/MWhth)	0,057	0,038	0,025	0,057	0,038	0,025	0,057	0,038	0,025
Variable costs (€/MWhth)	0,049	0,049	0,049	0,152	0,152	0,152	0,303	0,303	0,303
Fixed costs (€/MWhth)	0,015	0,010	0,007	0,015	0,010	0,007	0,015	0,010	0,007
Annual costs (€/MWhth)	0,121	0,097	0,081	0,224	0,200	0,184	0,375	0,351	0,335
Increase in electricity price (cent/kWhe)	0,032	0,026	0,021	0,059	0,053	0,048	0,099	0,092	0,088
Cost effciency (€/kgHg)	31633	25357	21174	21915	19563	17995	26636	24926	23785

Existing flue gas cleaning configuration: FF

	Emission limit: 1 µg/Nm3		
MERCURY EMISSIONS CALCULATION	**263 MW**	**658 MW**	**1316 MW**
POWER PLANT PROPERTIES			
Electrical power (MW)	100	250	500
Thermal power (MW)	263	658	1316
Fuel consumption (t/a)	249307	623269	1246537
FLUE GAS			
Flue gas volume-to-mass ratio (Nm3/kg fuel) (dry)	8,507	8,507	8,507
Flue gas volume flow (Nm3/h) (dry)	353476	883691	1767382
MERCURY REMOVAL			
MERCURY INPUT			
Hg input (kg/a)	34,65	86,63	173,27
Hg input (µg/Nm3) (dry, 6% O2)	17,14	17,14	17,14
MERCURY REMOVAL IN EXISTING EQUIPMENT			
Baseline Hg removal efficiency (%)	89,0 %	89,0 %	89,0 %
Baseline Hg removal (kg/a)	30,84	77,10	154,21
Baseline Hg removal (µg/Nm3) (dry, 6% O2)	15,25	15,25	15,25
Baseline Hg emissions (kg/a)	3,81	9,53	19,06
Baseline Hg emissions (µg/Nm3) (dry, 6% O2)	1,9	1,9	1,9
Hg emission limit (µg/Nm3)	1	1	1
ADDITIONAL MERCURY REMOVAL			
Required additional removal (%)	47,0 %	47,0 %	47,0 %
Required additional removal (kg/a)	1,79	4,48	8,95
Required additional removal (µg/Nm3) (dry, 6% O2)	0,9	0,9	0,9
TOTAL MERCURY REMOVAL			
Total Hg removal (%)	94,2 %	94,2 %	94,2 %
Total Hg removal (kg/a)	32,63	81,58	163,16
Total Hg removal (µg/Nm3) (dry, 6% O2)	16,14	16,14	16,14
MERCURY STACK EMISSIONS			
Hg stack emissions (kg/a)	2,02	5,05	10,11
Hg stack emissions (µg/Nm3) (dry, 6% O2)	1,0	1,0	1,0
COST-ANALYSIS	**263 MW**	**658 MW**	**1316 MW**
INVESTMENT COSTS			
Total investment cost (€/kWe)	8,01	5,34	3,56
Total investment cost (€)	**801 000**	**1 335 000**	**1 780 000**
CONSUMPTION RATES			
Consumption of the sorbent (mg/Nm3)	5	5	5
Consumption of the sorbent (kg/h)	1,8	4,6	9,2
Electricity (kW)	6,50	16,20	32,40
Electricity (MWh/a)	39,00	97,20	194,40
OPERATING COSTS			
Variable operating costs			
Sorbent (€/a)	19 597	48 992	97 984
Electricity (€/a)	3 171	7 902	15 805
Total variable costs (€/a)	**22 767**	**56 894**	**113 789**
Fixed operating costs			
Operation and maintenance (%/investment cost)	3 %	3 %	3 %
Operation and maintenance (€/a)	24 030	40 050	53 400
Total fixed costs (€/a)	**24 030**	**40 050**	**53 400**
Total operating costs (€/a)	**46 797**	**96 944**	**167 189**
YEARLY COSTS			
Interest rate (%)	8 %	8 %	8 %
Lifetime of the investment (a)	15	15	15
Annual investment costs (€/a)	90048	150080	200106
Operating costs (€/a)	46797	96944	167189
TOTA YEARLY COSTS (€/a)	**136845**	**247024**	**367295**
NOMINAL COSTS			
Annual investment cost (€/MWhth)	0,057	0,038	0,025
Variable costs (€/MWhth)	0,014	0,014	0,014
Fixed costs (€/MWhth)	0,015	0,010	0,007
Annual costs (€/MWhth)	0,087	0,063	0,047
Increase in electricity price (cent/kWhe)	0,023	0,016	0,012
Cost effciency (€/kgHg)	76449	55200	41038

Existing flue gas cleaning configuration: ESP+WFGD

	Emission limit: 4 µg/Nm3			Emission limit: 1 µg/Nm3		
MERCURY EMISSIONS CALCULATION	**263 MW**	**658 MW**	**1316 MW**	**263 MW**	**658 MW**	**1316 MW**
POWER PLANT PROPERTIES						
Electrical power (MW)	100	250	500	100	250	500
Thermal power (MW)	263	658	1316	263	658	1316
Fuel consumption (t/a)	249307	623269	1246537	249307	623269	1246537
FLUE GAS						
Flue gas volume-to-mass ratio (Nm3/kg fuel) (dry)	8,507	8,507	8,507	8,507	8,507	8,507
Flue gas volume flow (Nm3/h) (dry)	353476	883691	1767382	353476	883691	1767382
MERCURY REMOVAL						
MERCURY INPUT						
Hg input (kg/a)	34,65	86,63	173,27	34,65	86,63	173,27
Hg input (µg/Nm3) (dry, 6% O2)	17,14	17,14	17,14	17,14	17,14	17,14
MERCURY REMOVAL IN EXISTING EQUIPMENT						
Baseline Hg removal efficiency (%)	69,0 %	69,0 %	69,0 %	69,0 %	69,0 %	69,0 %
Baseline Hg removal (kg/a)	23,91	59,78	119,56	23,91	59,78	119,56
Baseline Hg removal (µg/Nm3) (dry, 6% O2)	11,83	11,83	11,83	11,83	11,83	11,83
Baseline Hg emissions (kg/a)	1,79	4,48	8,95	1,79	4,48	8,95
Baseline Hg emissions (µg/Nm3) (dry, 6% O2)	5,3	5,3	5,3	5,3	5,3	5,3
Hg emission limit (µg/Nm3)	4	4	4	1	1	1
ADDITIONAL MERCURY REMOVAL						
Required additional removal (%)	24,7 %	24,7 %	24,7 %	81,2 %	81,2 %	81,2 %
Required additional removal (kg/a)	2,66	6,64	13,28	8,72	21,80	43,60
Required additional removal (µg/Nm3) (dry, 6% O2)	1,3	1,3	1,3	4,3	4,3	4,3
TOTAL MERCURY REMOVAL						
Total Hg removal (%)	76,7 %	76,7 %	76,7 %	94,2 %	94,2 %	94,2 %
Total Hg removal (kg/a)	26,57	66,42	132,83	32,63	81,58	163,16
Total Hg removal (µg/Nm3) (dry, 6% O2)	13,14	13,14	13,14	16,14	16,14	16,14
MERCURY STACK EMISSIONS						
Hg stack emissions (kg/a)	8,09	20,22	40,44	2,02	5,05	10,11
Hg stack emissions (µg/Nm3) (dry, 6% O2)	4,0	4,0	4,0	1,0	1,0	1,0
COST-ANALYSIS	**263 MW**	**658 MW**	**1316 MW**	**263 MW**	**658 MW**	**1316 MW**
INVESTMENT COSTS						
Total investment cost (€/kWe)	8,01	5,34	3,56	90,5	90,5	90,5
Total investment cost (€)	**801 000**	**1 335 000**	**1 780 000**	**801 000**	**1 335 000**	**1 780 000**
CONSUMPTION RATES						
Consumption of the sorbent (mg/Nm3)	18,9	18,9	18,9	265	265	265
Consumption of the sorbent (kg/h)	6,9	17,3	34,7	33,2	83,0	166,1
Electricity (kW)	6,50	16,20	32,40	6,50	16,20	32,40
Electricity (MWh/a)	39,00	97,20	194,40	39,00	97,20	194,40
OPERATING COSTS						
Variable operating costs						
Sorbent (€/a)	74 076	185 189	370 379	354 701	886 754	1 773 507
Electricity (€/a)	3 171	7 902	15 805	3 171	7 902	15 805
Total variable costs (€/a)	**77 246**	**193 092**	**386 184**	**357 872**	**894 656**	**1 789 312**
Fixed operating costs						
Operation and maintenance (%/investment cost)	3 %	3 %	3 %	3 %	3 %	3 %
Operation and maintenance (€/a)	24 030	40 050	53 400	24 030	40 050	53 400
Total fixed costs (€/a)	**24 030**	**40 050**	**53 400**	**24 030**	**40 050**	**53 400**
Total operating costs (€/a)	**101 276**	**233 142**	**439 584**	**381 902**	**934 706**	**1 842 712**
YEARLY COSTS						
Interest rate (%)	8 %	8 %	8 %	8 %	8 %	8 %
Lifetime of the investment (a)	15	15	15	15	15	15
Annual investment costs (€/a)	90048	150080	200106	90048	150080	200106
Operating costs (€/a)	101276	233142	439584	381902	934706	1842712
TOTA YEARLY COSTS (€/a)	**191324**	**383222**	**639690**	**471950**	**1084786**	**2042818**
NOMINAL COSTS						
Annual investment cost (€/MWhth)	0,057	0,038	0,025	0,057	0,038	0,025
Variable costs (€/MWhth)	0,049	0,049	0,049	0,227	0,227	0,227
Fixed costs (€/MWhth)	0,015	0,010	0,007	0,015	0,010	0,007
Annual costs (€/MWhth)	0,121	0,097	0,081	0,299	0,275	0,259
Increase in electricity price (cent/kWhe)	0,032	0,026	0,021	0,079	0,072	0,068
Cost effciency (€/kgHg)	72059	57733	48185	54118	49756	46849

Existing flue gas cleaning configuration: ESP+SDA+FF

	Emission limit: 1 µg/Nm3		
MERCURY EMISSIONS CALCULATION	**263 MW**	**658 MW**	**1316 MW**
POWER PLANT PROPERTIES			
Electrical power (MW)	100	250	500
Thermal power (MW)	263	658	1316
Fuel consumption (t/a)	249307	623269	1246537
FLUE GAS			
Flue gas volume-to-mass ratio (Nm3/kg fuel) (dry)	8,507	8,507	8,507
Flue gas volume flow (Nm3/h) (dry)	353476	883691	1767382
MERCURY REMOVAL			
MERCURY INPUT			
Hg input (kg/a)	34,65	86,63	173,27
Hg input (µg/Nm3) (dry, 6% O2)	17,14	17,14	17,14
MERCURY REMOVAL IN EXISTING EQUIPMENT			
Baseline Hg removal efficiency (%)	93,0 %	93,0 %	93,0 %
Baseline Hg removal (kg/a)	32,23	80,57	161,14
Baseline Hg removal (µg/Nm3) (dry, 6% O2)	15,94	15,94	15,94
Baseline Hg emissions (kg/a)	2,43	6,06	12,13
Baseline Hg emissions (µg/Nm3) (dry, 6% O2)	1,2	1,2	1,2
Hg emission limit (µg/Nm3)	1	1	1
ADDITIONAL MERCURY REMOVAL			
Required additional removal (%)	16,6 %	16,6 %	16,6 %
Required additional removal (kg/a)	0,40	1,01	2,02
Required additional removal (µg/Nm3) (dry, 6% O2)	0,2	0,2	0,2
TOTAL MERCURY REMOVAL			
Total Hg removal (%)	94,2 %	94,2 %	94,2 %
Total Hg removal (kg/a)	32,63	81,58	163,16
Total Hg removal (µg/Nm3) (dry, 6% O2)	16,14	16,14	16,14
MERCURY STACK EMISSIONS			
Hg stack emissions (kg/a)	2,02	5,05	10,11
Hg stack emissions (µg/Nm3) (dry, 6% O2)	1,0	1,0	1,0
COST-ANALYSIS	**263 MW**	**658 MW**	**1316 MW**
INVESTMENT COSTS			
Total investment cost (€/kWe)	8,01	5,34	3,56
Total investment cost (€)	**801 000**	**1 335 000**	**1 780 000**
CONSUMPTION RATES			
Consumption of the sorbent (mg/Nm3)	5	5	5
Consumption of the sorbent (kg/h)	1,8	4,6	9,2
Electricity (kW)	6,50	16,20	32,40
Electricity (MWh/a)	39,00	97,20	194,40
OPERATING COSTS			
Variable operating costs			
Sorbent (€/a)	19 597	48 992	97 984
Electricity (€/a)	3 171	7 902	15 805
Total variable costs (€/a)	**22 767**	**56 894**	**113 789**
Fixed operating costs			
Operation and maintenance (%/investment cost)	3 %	3 %	3 %
Operation and maintenance (€/a)	24 030	40 050	53 400
Total fixed costs (€/a)	**24 030**	**40 050**	**53 400**
Total operating costs (€/a)	**46 797**	**96 944**	**167 189**
YEARLY COSTS			
Interest rate (%)	8 %	8 %	8 %
Lifetime of the investment (a)	15	15	15
Annual investment costs (€/a)	90048	150080	200106
Operating costs (€/a)	46797	96944	167189
TOTA YEARLY COSTS (€/a)	**136845**	**247024**	**367295**
NOMINAL COSTS			
Annual investment cost (€/MWhth)	0,057	0,038	0,025
Variable costs (€/MWhth)	0,014	0,014	0,014
Fixed costs (€/MWhth)	0,015	0,010	0,007
Annual costs (€/MWhth)	0,087	0,063	0,047
Increase in electricity price (cent/kWhe)	0,023	0,016	0,012
Cost effciency (€/kgHg)	338830	244653	181885

Existing flue gas cleaning configuration: SCR+ESP+WFGD

	Emission limit: 1 µg/Nm3		
MERCURY EMISSIONS CALCULATION	**263 MW**	**658 MW**	**1316 MW**
POWER PLANT PROPERTIES			
Electrical power (MW)	100	250	500
Thermal power (MW)	263	658	1316
Fuel consumption (t/a)	249307	623269	1246537
FLUE GAS			
Flue gas volume-to-mass ratio (Nm3/kg fuel) (dry)	8,507	8,507	8,507
Flue gas volume flow (Nm3/h) (dry)	353476	883691	1767382
MERCURY REMOVAL			
MERCURY INPUT			
Hg input (kg/a)	34,65	86,63	173,27
Hg input (µg/Nm3) (dry, 6% O2)	17,14	17,14	17,14
MERCURY REMOVAL IN EXISTING EQUIPMENT			
Baseline Hg removal efficiency (%)	85,0 %	85,0 %	85,0 %
Baseline Hg removal (kg/a)	29,46	73,64	147,28
Baseline Hg removal (µg/Nm3) (dry, 6% O2)	14,57	14,57	14,57
Baseline Hg emissions (kg/a)	5,20	13,00	25,99
Baseline Hg emissions (µg/Nm3) (dry, 6% O2)	2,6	2,6	2,6
Hg emission limit (µg/Nm3)	1	1	1
ADDITIONAL MERCURY REMOVAL			
Required additional removal (%)	61,1 %	61,1 %	61,1 %
Required additional removal (kg/a)	3,18	7,94	15,88
Required additional removal (µg/Nm3) (dry, 6% O2)	1,6	1,6	1,6
TOTAL MERCURY REMOVAL			
Total Hg removal (%)	94,2 %	94,2 %	94,2 %
Total Hg removal (kg/a)	32,63	81,58	163,16
Total Hg removal (µg/Nm3) (dry, 6% O2)	16,14	16,14	16,14
MERCURY STACK EMISSIONS			
Hg stack emissions (kg/a)	2,02	5,05	10,11
Hg stack emissions (µg/Nm3) (dry, 6% O2)	1,0	1,0	1,0
COST-ANALYSIS	**263 MW**	**658 MW**	**1316 MW**
INVESTMENT COSTS			
Total investment cost (€/kWe)	8,01	5,34	3,56
Total investment cost (€)	**801 000**	**1 335 000**	**1 780 000**
CONSUMPTION RATES			
Consumption of the sorbent (mg/Nm3)	51,8	51,8	51,8
Consumption of the sorbent (kg/h)	19,0	47,5	95,0
Electricity (kW)	6,50	16,20	32,40
Electricity (MWh/a)	39,00	97,20	194,40
OPERATING COSTS			
Variable operating costs			
Sorbent (€/a)	203 022	507 556	1 015 112
Electricity (€/a)	3 171	7 902	15 805
Total variable costs (€/a)	**206 193**	**515 459**	**1 030 917**
Fixed operating costs			
Operation and maintenance (%/investment cost)	3 %	3 %	3 %
Operation and maintenance (€/a)	24 030	40 050	53 400
Total fixed costs (€/a)	**24 030**	**40 050**	**53 400**
Total operating costs (€/a)	**230 223**	**555 509**	**1 084 317**
YEARLY COSTS			
Interest rate (%)	8 %	8 %	8 %
Lifetime of the investment (a)	15	15	15
Annual investment costs (€/a)	90048	150080	200106
Operating costs (€/a)	230223	555509	1084317
TOTA YEARLY COSTS (€/a)	**320271**	**705588**	**1284423**
NOMINAL COSTS			
Annual investment cost (€/MWhth)	0,057	0,038	0,025
Variable costs (€/MWhth)	0,131	0,131	0,131
Fixed costs (€/MWhth)	0,015	0,010	0,007
Annual costs (€/MWhth)	0,203	0,179	0,163
Increase in electricity price (cent/kWhe)	0,053	0,047	0,043
Cost effciency (€/kgHg)	100835	88860	80879

Appendix 4: Calculations for bromine injection

Existing flue gas cleaning configuration: ESP+WFGD

	Emission limit: 4 µg/Nm3			Emission limit: 1 µg/Nm3		
MERCURY EMISSIONS CALCULATION	**263 MW**	**658 MW**	**1316 MW**	**263 MW**	**658 MW**	**1316 MW**
POWER PLANT PROPERTIES						
Electrical power (MW)	100	250	500	100	250	500
Thermal power (MW)	263	658	1316	263	658	1316
Fuel consumption (t/a)	249307	623269	1246537	249307	623269	1246537
FLUE GAS						
Flue gas volume-to-mass ratio (Nm3/kg fuel) (dry)	8,507	8,507	8,507	8,507	8,507	8,507
Flue gas volume flow (Nm3/h) (dry)	353476	883691	1767382	353476	883691	1767382
MERCURY REMOVAL						
MERCURY INPUT						
Hg input (kg/a)	34,65	86,63	173,27	34,65	86,63	173,27
Hg input (µg/Nm3) (dry, 6% O2)	17,14	17,14	17,14	17,14	17,14	17,14
MERCURY REMOVAL IN EXISTING EQUIPMENT						
Baseline Hg removal efficiency (%)	69,0 %	69,0 %	69,0 %	69,0 %	69,0 %	69,0 %
Baseline Hg removal (kg/a)	23,91	59,78	119,56	23,91	59,78	119,56
Baseline Hg removal (µg/Nm3) (dry, 6% O2)	11,83	11,83	11,83	11,83	11,83	11,83
Baseline Hg emissions (kg/a)	1,79	4,48	8,95	1,79	4,48	8,95
Baseline Hg emissions (µg/Nm3) (dry, 6% O2)	5,3	5,3	5,3	5,3	5,3	5,3
Hg emission limit (µg/Nm3)	4	4	4	1	1	1
ADDITIONAL MERCURY REMOVAL						
Required additional removal (%)	24,7 %	24,7 %	24,7 %	81,2 %	81,2 %	81,2 %
Required additional removal (kg/a)	2,66	6,64	13,28	8,72	21,80	43,60
Required additional removal (µg/Nm3) (dry, 6% O2)	1,3	1,3	1,3	4,3	4,3	4,3
TOTAL MERCURY REMOVAL						
Total Hg removal (%)	76,7 %	76,7 %	76,7 %	94,2 %	94,2 %	94,2 %
Total Hg removal (kg/a)	26,57	66,42	132,83	32,63	81,58	163,16
Total Hg removal (µg/Nm3) (dry, 6% O2)	13,14	13,14	13,14	16,14	16,14	16,14
MERCURY STACK EMISSIONS						
Hg stack emissions (kg/a)	8,09	20,22	40,44	2,02	5,05	10,11
Hg stack emissions (µg/Nm3) (dry, 6% O2)	4,0	4,0	4,0	1,0	1,0	1,0
COST-ANALYSIS	**263 MW**	**658 MW**	**1316 MW**	**263 MW**	**658 MW**	**1316 MW**
INVESTMENT COSTS						
Total investment cost (€/kWe)	4,62	2,67	1,76	4,62	2,67	1,76
Total investment cost (€)	**462 000**	**667 500**	**880 000**	**462 000**	**667 500**	**880 000**
CONSUMPTION RATES						
Consumption of the bromine (mg/kg fuel)	25	25	25	150	150	150
Consumption of the bromine (kg/h)	1,0	2,6	5,2	6,2	15,6	31,2
Electricity (kW)	-	-	-	-	-	-
Electricity (MWh/a)	-	-	-	-	-	-
OPERATING COSTS						
Variable operating costs						
Calcium bromide 52wt% (€/a)	29 385	73 462	146 923	176 308	440 770	881 540
Electricity (€/a)	-	-	-	-	-	-
Total variable costs (€/a)	**29 385**	**73 462**	**146 923**	**176 308**	**440 770**	**881 540**
Fixed operating costs						
Operation and maintenance (%/investment cost)	3 %	3 %	3 %	3 %	3 %	3 %
Operation and maintenance (€/a)	13 860	20 025	26 400	13 860	20 025	26 400
Total fixed costs (€/a)	**13 860**	**20 025**	**26 400**	**13 860**	**20 025**	**26 400**
Total operating costs (€/a)	**43 245**	**93 487**	**173 323**	**190 168**	**460 795**	**907 940**
YEARLY COSTS						
Interest rate (%)	8 %	8 %	8 %	8 %	8 %	8 %
Lifetime of the investment (a)	15	15	15	15	15	15
Annual investment costs (€/a)	51938	75040	98929	51938	75040	98929
Operating costs (€/a)	43245	93487	173323	190168	460795	907940
TOTA YEARLY COSTS (€/a)	**95182**	**168527**	**272252**	**242106**	**535835**	**1006869**
NOMINAL COSTS						
Annual investment cost (€/MWhth)	0,033	0,019	0,013	0,033	0,019	0,013
Variable costs (€/MWhth)	0,019	0,019	0,019	0,112	0,112	0,112
Fixed costs (€/MWhth)	0,009	0,005	0,003	0,009	0,005	0,003
Annual costs (€/MWhth)	0,060	0,043	0,034	0,153	0,136	0,128
Increase in electricity price (cent/kWhe)	0,016	0,011	0,009	0,040	0,036	0,034
Cost effciency (€/kgHg)	35849	25389	20508	27762	24577	23091

Existing flue gas cleaning configuration: ESP+SDA+FF

	Emission limit: 1 µg/Nm3		
MERCURY EMISSIONS CALCULATION	**263 MW**	**658 MW**	**1316 MW**
POWER PLANT PROPERTIES			
Electrical power (MW)	100	250	500
Thermal power (MW)	263	658	1316
Fuel consumption (t/a)	249307	623269	1246537
FLUE GAS			
Flue gas volume-to-mass ratio (Nm3/kg fuel) (dry)	8,507	8,507	8,507
Flue gas volume flow (Nm3/h) (dry)	353476	883691	1767382
MERCURY REMOVAL			
MERCURY INPUT			
Hg input (kg/a)	34,65	86,63	173,27
Hg input (µg/Nm3) (dry, 6% O2)	17,14	17,14	17,14
MERCURY REMOVAL IN EXISTING EQUIPMENT			
Baseline Hg removal efficiency (%)	93,0 %	93,0 %	93,0 %
Baseline Hg removal (kg/a)	32,23	80,57	161,14
Baseline Hg removal (µg/Nm3) (dry, 6% O2)	15,94	15,94	15,94
Baseline Hg emissions (kg/a)	2,43	6,06	12,13
Baseline Hg emissions (µg/Nm3) (dry, 6% O2)	1,2	1,2	1,2
Hg emission limit (µg/Nm3)	1	1	1
ADDITIONAL MERCURY REMOVAL			
Required additional removal (%)	16,6 %	16,6 %	16,6 %
Required additional removal (kg/a)	0,40	1,01	2,02
Required additional removal (µg/Nm3) (dry, 6% O2)	0,2	0,2	0,2
TOTAL MERCURY REMOVAL			
Total Hg removal (%)	94,2 %	94,2 %	94,2 %
Total Hg removal (kg/a)	32,63	81,58	163,16
Total Hg removal (µg/Nm3) (dry, 6% O2)	16,14	16,14	16,14
MERCURY STACK EMISSIONS			
Hg stack emissions (kg/a)	2,02	5,05	10,11
Hg stack emissions (µg/Nm3) (dry, 6% O2)	1,0	1,0	1,0
COST-ANALYSIS	**263 MW**	**658 MW**	**1316 MW**
INVESTMENT COSTS			
Total investment cost (€/kWe)	4,62	2,67	1,76
Total investment cost (€)	**462 000**	**667 500**	**880 000**
CONSUMPTION RATES			
Consumption of the bromine (mg/kg fuel)	25	25	25
Consumption of the bromine (kg/h)	1,0	2,6	5,2
Electricity (kW)	-	-	-
Electricity (MWh/a)	-	-	-
OPERATING COSTS			
Variable operating costs			
Calcium bromide 52wt% (€/a)	29 385	73 462	146 923
Electricity (€/a)	-	-	-
Total variable costs (€/a)	**29 385**	**73 462**	**146 923**
Fixed operating costs			
Operation and maintenance (%/investment cost)	3 %	3 %	3 %
Operation and maintenance (€/a)	13 860	20 025	26 400
Total fixed costs (€/a)	**13 860**	**20 025**	**26 400**
Total operating costs (€/a)	**43 245**	**93 487**	**173 323**
YEARLY COSTS			
Interest rate (%)	8 %	8 %	8 %
Lifetime of the investment (a)	15	15	15
Annual investment costs (€/a)	51938	75040	98929
Operating costs (€/a)	43245	93487	173323
TOTA YEARLY COSTS (€/a)	**95182**	**168527**	**272252**
NOMINAL COSTS			
Annual investment cost (€/MWhth)	0,033	0,019	0,013
Variable costs (€/MWhth)	0,019	0,019	0,019
Fixed costs (€/MWhth)	0,009	0,005	0,003
Annual costs (€/MWhth)	0,060	0,043	0,034
Increase in electricity price (cent/kWhe)	0,016	0,011	0,009
Cost effciency (€/kgHg)	235672	166909	134820

Existing flue gas cleaning configuration: SCR+ESP+WFG

	Emission limit: 1 µg/Nm3		
MERCURY EMISSIONS CALCULATION	**263 MW**	**658 MW**	**1316 MW**
POWER PLANT PROPERTIES			
Electrical power (MW)	100	250	500
Thermal power (MW)	263	658	1316
Fuel consumption (t/a)	249307	623269	1246537
FLUE GAS			
Flue gas volume-to-mass ratio (Nm3/kg fuel) (dry)	8,507	8,507	8,507
Flue gas volume flow (Nm3/h) (dry)	353476	883691	1767382
MERCURY REMOVAL			
MERCURY INPUT			
Hg input (kg/a)	34,65	86,63	173,27
Hg input (µg/Nm3) (dry, 6% O2)	17,14	17,14	17,14
MERCURY REMOVAL IN EXISTING EQUIPMENT			
Baseline Hg removal efficiency (%)	85,0 %	85,0 %	85,0 %
Baseline Hg removal (kg/a)	29,46	73,64	147,28
Baseline Hg removal (µg/Nm3) (dry, 6% O2)	14,57	14,57	14,57
Baseline Hg emissions (kg/a)	5,20	13,00	25,99
Baseline Hg emissions (µg/Nm3) (dry, 6% O2)	2,6	2,6	2,6
Hg emission limit (µg/Nm3)	1	1	1
ADDITIONAL MERCURY REMOVAL			
Required additional removal (%)	61,1 %	61,1 %	61,1 %
Required additional removal (kg/a)	3,18	7,94	15,88
Required additional removal (µg/Nm3) (dry, 6% O2)	1,6	1,6	1,6
TOTAL MERCURY REMOVAL			
Total Hg removal (%)	94,2 %	94,2 %	94,2 %
Total Hg removal (kg/a)	32,63	81,58	163,16
Total Hg removal (µg/Nm3) (dry, 6% O2)	16,14	16,14	16,14
MERCURY STACK EMISSIONS			
Hg stack emissions (kg/a)	2,02	5,05	10,11
Hg stack emissions (µg/Nm3) (dry, 6% O2)	1,0	1,0	1,0
COST-ANALYSIS	**263 MW**	**658 MW**	**1316 MW**
INVESTMENT COSTS			
Total investment cost (€/kWe)	4,62	2,67	1,76
Total investment cost (€)	**462 000**	**667 500**	**880 000**
CONSUMPTION RATES			
Consumption of the bromine (mg/kg fuel)	25	25	25
Consumption of the bromine (kg/h)	1,0	2,6	5,2
Electricity (kW)	-	-	-
Electricity (MWh/a)	-	-	-
OPERATING COSTS			
Variable operating costs			
Calcium bromide 52wt% (€/a)	29 385	73 462	146 923
Electricity (€/a)	-	-	-
Total variable costs (€/a)	**29 385**	**73 462**	**146 923**
Fixed operating costs			
Operation and maintenance (%/investment cost)	3 %	3 %	3 %
Operation and maintenance (€/a)	13 860	20 025	26 400
Total fixed costs (€/a)	**13 860**	**20 025**	**26 400**
Total operating costs (€/a)	**43 245**	**93 487**	**173 323**
YEARLY COSTS			
Interest rate (%)	8 %	8 %	8 %
Lifetime of the investment (a)	15	15	15
Annual investment costs (€/a)	51938	75040	98929
Operating costs (€/a)	43245	93487	173323
TOTA YEARLY COSTS (€/a)	**95182**	**168527**	**272252**
NOMINAL COSTS			
Annual investment cost (€/MWhth)	0,033	0,019	0,013
Variable costs (€/MWhth)	0,019	0,019	0,019
Fixed costs (€/MWhth)	0,009	0,005	0,003
Annual costs (€/MWhth)	0,060	0,043	0,034
Increase in electricity price (cent/kWhe)	0,016	0,011	0,009
Cost effciency (€/kgHg)	29968	21224	17143